Section 4
Analysis Phase

This phase of the ERA analyzes exposure data (Exposure Assessment) and effects data (Effects Assessment) for the major stressors (PCBs) and representative receptors previously identified in Problem Formulation.

4.1 Ecological Exposure Assessment

Exposure Assessment evaluates and summarizes available exposure data, including exposure-related data on potential ecological receptors. The primary output of exposure assessment is an exposure profile that presents the magnitude (e.g., concentration) and distribution (e.g., surface water, sediment) of stressors to which ecological receptors may be exposed. For this ERA, the primary chemical stressors are PCBs because of the magnitude and extent of PCB contamination onsite. This focused ERA recognizes that other potential chemical stressors have been identified in the environment, but considers these other chemical stressors to be of much less ecological concern (i.e., much lower risk) than PCBs. Exposure profiles serve as input into the final stage of risk assessment, Risk Characterization.

4.1.1 Exposure Profiles – PCBs

Exposure Profiles describe the magnitude and distribution of stressors identified in the Problem Formulation phase. Exposure profiles for PCBs are summarized in Tables 4-1 and 4-2. Table 4-1 includes the sitewide range of total PCB concentrations and identifies the individual Aroclors for which abiotic media were sampled. Table 4-2 includes summary data on important chemical properties (i.e., environmental persistence, bioavailability, and bioconcentration potential) for PCBs. Non-chemical stressors are discussed in Section 4.1.2.

Recently collected data considered useable for risk assessment purposes are used to describe the magnitude and distribution of PCBs in the API/PC/KR environment. The majority of the abiotic (i.e., sediment, water, surface soil) data used in this ERA are from 1993 and 1994, when most of the biological sampling was conducted. Some floodplain sediment/soil samples collected during this time period were achieved under stable conditions and analyzed in 1997. The floodplain sediment/soil database used in this ERA is based on data from samples collected in 1993 and 1994, including those analyzed in 1997. Where data gaps have been identified, they have been addressed with data collected before 1993 and rarely after 1994. For example, data on PCB concentrations in plants were collected in 2000. In nearly all cases where pre-1993 were used, they were taken from the Description of the Current Situation (BBL 1992). With the exception noted above, data collected since 1994 are not included in the ERA because it is important to compare abiotic and biological data from the same time period to the extent possible. The extensive aquatic and terrestrial biological sampling conducted in 1993 serves as the basis for this ERA. Abiotic data collected in 1993 and 1994 are therefore considered most useful for comparison purposes. Such data are used in this ERA except where important data gaps are identified. The

relationships between biological data and abiotic data are established or estimated only for those ABSAs associated with 1993/1994 data. Where such data are lacking for a location or an abiotic media type, relationships are not established. These relationships include the derivation of soil/water partition factors, bioconcentration factors (BCFs), and biota-sediment accumulation factors (BSAFs).

Although no single concentration value can truly represent the variability of chemical concentrations measured in each medium of concern, the arithmetic mean value best represents the average concentration to which API/PC/KR receptors may be exposed. Where sufficient data have been collected, the arithmetic mean represents the average exposure concentration and the upper 95th confidence limit of the arithmetic mean (U95) is often used to represent a reasonable maximum exposure. Support for using U95 values is found in EPA guidance (1992b) for calculating values that are most representative of the higher end of actual chemical concentrations in environmental media to which human or ecological receptors may be exposed. This guidance states, however, that calculation of U95 values is appropriate only when sufficient data are available. In some cases, insufficient data have been collected from each individual sampling location to allow for complete confidence in U95 values. In cases where data are minimal, calculated U95 values sometimes exceed maximum detected concentrations.

Sufficient data for calculating U95 values have been collected for most abiotic and some biological media (e.g., fish). U95 values are therefore used to represent exposure concentrations in abiotic media and for those biological data associated with sufficient data. The latter category includes whole body fish data. Arithmetic mean and maximum PCB concentrations in most media are also presented in this section for comparison purposes. Arithmetic means include non-detect (ND) data using two accepted methods based on the source of the data. Means of abiotic data collected in 1994 are derived using a randomly selected number between zero and the laboratory reported detection limit to represent non-detects. In the few cases where older abiotic data are used, means are derived using the EPA-recommended method where half the detection limit is used to represent non-detects.

In cases where data are insufficient for deriving confident U95 values (e.g., mink, earthworms, mouse, and muskrat), maximum detected values are used because they probably best represent reasonable maximum exposures. This is especially true where, because data are limited, the true maximum exposure concentrations are unlikely to have been measured. This approach is scientifically defensible considering data limitations, and in fact follows guidance provided by state and federal regulatory agencies. For the most part, however, U95 values are considered representative of reasonable maximum exposure concentrations and are preferred where data quantity allows confidence in the derived values.

Finally, because this ERA is not based on a single line of evidence or single exposure point concentrations, the distribution of potential exposure concentrations associated

with abiotic media is also considered important. For this reason, the arithmetic mean, U95, and maximum concentration of PCBs in abiotic media are also compared to relevant effects concentrations to additionally describe risks. These descriptions are presented graphically in Section 5 (Risk Characterization) for PCBs in surface water, streambed and floodplain sediment, and surface soil for each of the defined sampling areas. These graphical presentations (Figures 5-1 to 5-4) present total PCB concentrations for each abiotic media type overlaid with relevant media-specific effects concentrations, criteria, or thresholds.

Table 4-1 presents the sitewide (non-reference) and reference area ranges of total PCB concentrations detected in abiotic media. Table 4-2 presents important chemical properties for the PCBs identified at the API/PC/KR. Each of these properties is discussed below.

Environmental Persistence

Environmental persistence indicates whether a chemical is likely to be long-lasting in the environment or, alternatively, be degraded by natural processes. Higher chlorinated PCBs, i.e., those with five or more chlorine atoms, are more persistent in the environment than those with three or less chlorine atoms (Eisler 1986). PCBs in sediments (including floodplain sediments) at the API/PC/KR site are the higher chlorinated Aroclors.

Bioconcentration Potential

Bioconcentration potential indicates whether a chemical is likely to be retained in biological tissues after it is taken in by ingestion or other means. Retention of chemicals is not in itself an appropriate measurement endpoint unless it is associated with adverse ecological effects. Retention is, however, useful for verifying exposure and for evaluating bioavailability and the potential for food chain/food web effects. BCFs, derived under equilibrium conditions, are often used as screening-level data to evaluate bioconcentration potential. BCFs are based on the ratio of contaminant concentration in aquatic biota to contaminant concentration in water. Because BCFs are derived under equilibrium conditions and under relatively long exposure durations, they consider both uptake and elimination (depuration) rates. Chemicals with BCFs greater than 300 generally indicate a potential to bioconcentrate (EPA 1991). Chemicals with log BCFs above 3 (BCFs above 1,000) are considered to have significant potential to bioaccumulate (EPA 1992b). For this ERA, available freshwater BCFs for invertebrates and fish that have potential to occur in the API/PC/KR site, or those that are closely related to indigenous species, are used to evaluate bioconcentration potential. In addition, degree of chlorination for individual Aroclors is commonly used to estimate bioconcentration potential.

Bioavailability

For this ERA, bioavailable chemicals are defined as those that exist in a form that has the ability to cause adverse ecological effects or bioaccumulate. As stated previously, bioaccumulation may not in itself constitute a significant ecological effect, but

provides important evidence of both exposure and potential for causing adverse effects to multiple trophic levels under certain conditions. For example, some lipophilic chemicals, such as PCBs, are taken up by biota and are stored in fatty tissues with no apparent ill effects. However, under stressful conditions, such as during winter when only poor quality foods are available, these fats are metabolized and the contaminants can then cause adverse effects.

Chemical properties (e.g., degree of chlorination) or environmental conditions (e.g., high levels of dissolved and particulate organic carbon) can affect the potential bioavailability and toxicity of many chemicals, including PCBs. The bioavailability and, therefore, toxicity of some PCBs in surface water can be influenced by the concentration of dissolved organic carbon. In addition, sediment organic carbon content, measured as total organic carbon (TOC), apparently affects bioavailability and toxicity of some PCBs. For some chemicals, chemical form and thus toxicity can change rather rapidly under changing environmental conditions (e.g., fluctuations in pH, temperature, or surface water flow). Seasonal conditions such as snowmelt and rainfall are likely to affect bioavailability of PCBs in the API/PC/KR. For the most part, however, PCB bioavailability (and potential toxicity) is expected to remain fairly stable because PCBs bind strongly to organic particulate matter. Once taken up by animals, PCBs are likely to be stored predominately in fatty tissues. PCB analyses of biological tissues generally measure Aroclor 1254 and (especially) Aroclor 1260. This finding is supported by studies that show biological conversion of one Aroclor to another after uptake. The chemical mixtures found in abiotic exposure media show little resemblance to Aroclors measured in biological tissues (Eisler 1986). The finding that PCBs have been detected in the tissues of all sampled biota comprising multiple trophic levels at concentrations exceeding important thresholds supports the preliminary assumption that PCBs at this site are indeed bioavailable.

4.1.2 Exposure Profiles – Non-chemical Stressors

Although not the focus of Superfund risk assessments, non-chemical stressors such as disturbed habitats can also affect ecological receptors. Such stressors can therefore be important components of exposure profiles. Non-chemical stressors identified for the API/PC/KR include multiple impacts due to urbanized settings, and may include siltation of instream substrates, historical damming of Portage Creek and the Kalamazoo River, and disturbed riparian/terrestrial habitats adjacent to both the creek and the river. These physical stressors occur throughout the API/PC/KR site to limited degrees, but the extent and severity of such impacts are expected to minimal when compared to the wide ranging impacts of exposure to PCBs. The potential effects of these non-chemical stressors are discussed in Effects Characterization (Section 4.2) of the ERA.

4.1.3 Exposure Scenarios

Exposure-related information for each of the representative groups of organisms previously identified as potential receptors for this ERA is described in this section.

These descriptions are based on likely exposure scenarios preliminarily identified in the SCEM developed in the Problem Formulation phase of the ERA. These preliminary exposure scenarios are refined for the major representative receptors or receptor groups previously identified.

The receptor groups are represented by organisms identified in Section 3.2.3, and include those that are presently being exposed or have potential to be exposed under current conditions. Exposure scenarios, summarized in Table 4-3, are simplified descriptions of how potential receptors or representative receptor groups may come in contact with previously identified stressors.

As presented in Table 4-3, some organisms or representative groups of organisms can be exposed to contaminants by direct uptake (through or on roots of plants) or by ingestion of contaminated media and/or prey. Estimates of plant uptake are most appropriately based on site-specific soil-to-plant transfer factors for the specific plant species and tissues (e.g., fruits) likely to be consumed. Species-specific plant data are limited, however, and do not include a wide variety of plant species or tissues likely to be eaten by representative receptors such as mouse, muskrat, or fox. Daily intake rates for representative animals are most appropriately calculated using site-specific data (e.g., contaminant concentrations in food items and dietary composition). Site-specific data related to diet of consumers and certain other critical input parameters are, however, unavailable for this ERA. Daily intake rates for terrestrial animals are therefore based on literature values for dietary intake and site-specific tissue data where such data exist.

Although several potential exposure scenarios can be identified for ecological receptors, it is most appropriate to focus the assessment on critical exposure scenarios. This ERA is focused on the most critical exposure scenarios identified in the SCM (Figure 3-11). Critical exposure scenarios are discussed below.

Aquatic Exposures

The primary PCB-related risks for aquatic organisms are likely to be from direct contact with and ingestion of contaminated surface water (including suspended sediments) in areas where surface water PCB concentrations are elevated. In addition, ingestion of bottom sediment and sediment pore (interstitial) water with elevated PCBs poses risks to benthic invertebrates, bottom-dwelling fish, and to varying extents, other aquatic biota.

Finally, aquatic organisms that occupy upper trophic levels can be adversely affected by ingesting PCB-contaminated prey. The relative contribution from each exposure source (surface water, sediment, interstitial water, prey) to overall aquatic exposure to PCBs cannot, however, be reliably determined for most aquatic organisms because data describing the variability in factors that can affect total exposure are lacking. These factors can include intraspecific and interspecific differences in life stage, season, diet, ingestion rate, specific habitat, etc. This assessment evaluates potential

risks posed to aquatic biota primarily by comparing ambient PCB concentrations in surface water and streambed sediment to media-specific criteria, such as chronic ambient water quality criteria (AWQC) and critical effects concentrations (e.g., no or low observed adverse effects concentrations) for appropriate species.

Semi-Aquatic and Terrestrial Exposures

Because PCBs tend to bioconcentrate to a high degree and biomagnify, ingestion of contaminated surface water and surface soil by terrestrial animals is expected to be less significant than ingestion of contaminated food. The uptake of chemical contaminants by terrestrial plants can also be important if the contaminants of concern are easily taken up, phytotoxic, or can cause food chain effects to herbivorous consumers. The importance of the food-ingestion pathway and uptake by terrestrial plants depends, however, on the types and abundance of plant and animal receptors as well as on the types and concentrations of chemical contaminants present. Terrestrial/riparian wildlife are common along the API/PC/KR, even though riparian and terrestrial habitats have been visibly degraded in some areas. Significant potential, therefore, exists for terrestrial and riparian species to be exposed to PCB contamination.

Terrestrial/riparian plant communities along the API/PC/KR have been affected by past industrial activities and other human-induced stresses. In some areas containing PCB residual material (e.g., A-Site) the effects are sufficiently limiting to preclude the existence of vegetation, and in other areas existing plant communities are dominated by "weedy" type forbs and shrubs. The causes of observed stress on certain plant communities has not been determined, but may be the result of physical (e.g., habitat alteration) or chemical (contamination/toxicity) stress.

Most herbivorous wildlife species are unlikely to frequent the few barren areas observed; however, those areas dominated by weedy forbs may be an attraction to certain receptors within the API/PC/KR area. Several terrestrial/riparian vertebrate species common in western Michigan that require suitable vegetative cover and other specific habitat requirements (e.g., muskrat and white-footed mouse) are commonly observed within all or most portions of the API/PC/KR area. Although suitable habitat for mink is available throughout most of the API/PC/KR area, populations appear depressed based on mink trapping results.

Because vegetation is only rarely absent or visibly stressed within the API/PC/KR area, and because herbivorous wildlife are common, plant consumers can be exposed to site-related contaminants (e.g., PCBs) under present conditions. Similarly, most predators or consumers of herbivorous species can also be exposed to site-related contaminants because adequate cover and prey are generally available.

Although a large variety of commonly observed terrestrial animal species including resident and migratory birds have been reported onsite, certain other local types of animals species that are not easily observed or often reported probably also occur

regularly or permanently within the API/PC/KR area. These include macroinvertebrates (e.g., insects, spiders, centipedes, millipedes), amphibians (e.g., toads, Ranid frogs, tree frogs, salamanders, newts), reptiles (e.g., lizards, snakes, turtles), and mammals (e.g., shrews, raccoons, voles, skunks, weasels, etc.) and are summarized in the tables in Appendix A. Although for the most part data are lacking, risks to these organisms could occur as a result of direct contact with or ingestion of contaminants via surface water, sediment, soil, and food items. For many terrestrial ecological receptors exposed to PCBs, the most important pathway involves ingestion of PCB-contaminated prey. Finally, PCB exposures are likely to be limited in areas with insufficient cover and prey because such areas are probably avoided by most terrestrial species.

Portions of the API/PC/KR riparian habitat have been reduced by commercial, industrial, and residential development. Many resident species have apparently adapted to the encroachment of humans and these species can therefore be found in close proximity of the landfills and abandoned industrial facilities along the Kalamazoo River and Portage Creek.

Exposures via Food Chain Transfer

PCBs detected onsite have been in the environment for some time, and as a result are considered weathered. Weathered PCBs are comprised of various combinations of different PCB congeners that differ in their environmental persistence and toxicity. Most of the PCB data used in this ERA are based on Aroclor analyses, and exposures are described using total PCB data. PCBs are known to bioaccumulate as a result of ingestion of PCB-contaminated surface water, sediment, soil, vegetation, and prey. BCFs or bioaccumulation factors (BAFs) are often used to evaluate the bioaccumulation potential of chemicals in the environment. As stated previously, chemicals with BCFs less than 300 are considered to have low bioaccumulation potential, while those with BCF between 300 and 1,000 have moderate potential to bioaccumulate. Chemicals with BCFs greater than 1,000 are of most concern with regard to potential bioaccumulation. Table 4-2 lists literature-based freshwater BCFs for the PCBs detected onsite.

Upper trophic level predators, such as mink or bald eagle, are likely to be most exposed to PCBs via consumption of contaminated prey. Food webs for such species can be based on PCBs in surface soil, instream sediment, or floodplain sediment/soils. Bald eagles, for example, are most closely associated with PCBs in fish, which in turn are exposed to PCBs in the water column, instream sediments, and prey. For other species such as mink, dietary exposures are likely to be based on a variety of abiotic media, including surface water, instream sediment, floodplain sediment, and possibly surface soils in more upland areas. Food chain modeling requires that the relationships between source media and prey be known. Food chain modeling is used to calculate PCB doses and dose-based hazard quotients.

Media-specific preliminary remedial goals (PRGs) are also calculated using food chain modeling for most upper trophic level receptors except mink. PRGs for mink are based on the site-specific relationships between PCBs in fish, water, and sediment instead of on food chain modeling for the reasons discussed below.

(1) The inclusion of mixed terrestrial and aquatic prey means that two PRGs (soil and sediment) need to be solved simultaneously, which results in an array of possible combinations of protective soil and sediment PRGs.

(2) Since the experimental species and receptor species are the same, a simplified approach is permissible—(i.e., back-calculating PRGs from dietary PCB concentrations protective of mink, instead of the body weight normalized approach required for extrapolating toxicity information between species).

(3) The modeled terrestrial component of riverine mink diet is minimal (~15% of total diet), and the central question is *what level of sediment PCBs would be protective of mink predominately feeding on aquatic resources*.

4.1.4 Exposure Analysis

Information on distributions of stressors and receptors are combined and summarized in this section, and potential for exposure is discussed. For PCBs, such discussions consider important chemical properties summarized in Table 4-2 (i.e., environmental persistence, bioavailability, and bioconcentration potential). For identified receptors or representative groups of receptors, estimates of potential exposure consider the important ecological parameters that can increase or in other ways modify exposure, such as habitat use and foraging behavior. Exposure-related information for key organisms or representative receptors is summarized in Appendix B.

Samples of several representative organisms, including some of those discussed above, were collected and analyzed for whole body PCB analyses. The U95 (fish) and maximum (terrestrial biota) whole body PCB concentration for each of these organisms or groups of organisms is used to evaluate PCB exposure in representative biota, and support food chain modeling.

The concentrations and ABSA-wide distributions of PCBs in sampled biota and abiotic media are presented in Tables 4-5a and 4-5b.

Table 4-5a presents all other biological and abiotic concentration data. These data are presented on an area-by-area basis. This presentation is, for Table 4-5a, based on previously defined spatial units for sampling aquatic biota (ABSAs) and terrestrial biota (TBSAs) (Figures 3-1 to 3-10). As discussed previously, boundaries of ABSAs are defined so that all areas of the API/PC/KR site are associated with an ABSA. This expansion of ABSAs beyond sampled areas is not intended to suggest that the abiotic (i.e., sediment, soil, and water) samples collected are representative of non-sampled areas within the ABSA. The variability of such samples precludes having much

confidence in such assumptions. Instead, the ABSAs are expanded in consideration of mobile receptors such as fish and mink. The PCB concentrations of mobile receptors collected within an ABSA are assumed to be (1) representative of concentrations in mobile biota found in the expanded ABSA, and (2) the result of exposures from within the entire ABSA.

Table 4-5b presents total PCB concentrations measured in bird eggs collected onsite. In most cases these egg data include total PCB concentrations in individual eggs taken from the same nest. Where this is the case, these data cannot be considered completely independent samples because the eggs were laid by the same parent bird. Multiple eggs were taken from nests of most bird species listed in Table 4-5b.

Figure 4-1 graphically presents the relationships between PCBs in surface water, sediment, and whole body fish collected onsite, on an ABSA-specific basis. This figure reveals that PCB concentrations in fish and abiotic media are generally related but the relationship is not linear. This finding is not unexpected since fish receive PCBs from multiple sources and via several exposure pathways. PCB concentrations in fish tissue are therefore not expected to be completely correlated to PCB concentrations in surface water, sediment, or prey. More importantly, it is expected and confirmed that elevated fish tissue PCB concentrations are associated with elevated PCB concentrations in abiotic media. In addition, low fish tissue PCB concentrations are associated with low PCB concentrations in abiotic media.

4.1.5 Food Web/Food Chain Modeling

The PCB Food Web Model (Figure 4-2) is described below and food web-related data are presented in Appendices C-1 and C-2. Appendix C-1 presents the input parameters and concentration data for abiotic and biotic media. Appendix C-2 is a spreadsheet used to calculate doses and PRGs for representative semi-aquatic and terrestrial receptors.

This food web model is an important component of the ERA because it describes important characteristics of key receptors and associated exposures to PCBs. These key species were selected because they are common or potential inhabitants of the API/PC/KR corridor and most likely obtain their food from the river and/or associated terrestrial habitats. EPA Region 5 Biological Technical Advisory Group (BTAG) has approved these key species for this ERA. Section 5.1.4 provides a discussion on the estimated average potential daily dosage (APDD) and threshold effects values for "key" species. This is a simplified model utilizing measured and estimated input parameters and established mathematical relationships between input parameters. Models such as these are used to estimate the average potential dietary exposure for upper trophic level organisms from ingestion of contaminated prey. For this ERA, the risks posed to lower trophic level organisms and all aquatic organisms are assessed by comparing exposure point concentrations in exposure media to concentrations that can cause ecologically significant effects. For this ERA, ecologically significant effects are defined as those adversely affecting survival,

growth, or reproduction. Survival or mortality can be determined in acute toxicity tests (i.e., tests of short duration and generally high exposure concentrations) or chronic toxicity tests (i.e., tests of long duration and comparatively lower exposure concentrations). Growth and reproductive effects are usually measured by chronic testing.

PCBs are not acutely toxic to many species, yet long-term exposures can have adverse effects on individuals, populations, and communities. The presence of detectable PCB concentrations in biological tissues is not in itself considered ecologically significant unless such concentrations can be correlated to adverse effects. For example, common snapping turtles (*Chelydra serpentina*) are known to accumulate and retain substantial amounts of PCBs in fatty tissues with no observed ill effects (Olafsson, et al. 1983 in Eisler 1986). Consumers of snapping turtles, however, may be at significant risk if dietary intake is of sufficient quantity, frequency, and duration to result in exposure to PCB concentrations similar to those measured at the API/PC/KR site.

As previously stated, it is most appropriate to focus the ERA on critical exposure scenarios. This ERA, and specifically the food web model, is focused on the most critical exposure scenarios for ecological receptors. For terrestrial species, and for nearly all identified carnivores, the potential exposure from ingestion of PCB-contaminated surface water is considered insignificant relative to the potential risks from ingestion of PCB-contaminated prey. This assumption is based on relatively low surface water PCB concentrations and total potential PCB intake compared to prey concentrations and total potential intake via ingestion of contaminated prey. The risks to carnivores and all terrestrial species from the ingestion of PCB-contaminated surface water are, therefore, not included in this assessment.

The primary PCB-related risks for aquatic organisms, especially those occupying lower trophic levels, are likely to be from direct contact with and ingestion of contaminated surface water, sediment, and pore or interstitial water. Certain aquatic organisms such as predatory game fish can also be significantly exposed to PCBs through ingestion of contaminated prey. The relative contribution to overall PCB exposure from each exposure pathway and exposure source (e.g., water, sediment, prey) cannot, however, be reliably determined for most aquatic organisms because of the variability in factors that can affect total exposure.

These factors can include intraspecific and interspecific differences in life stage, season, diet, ingestion rate, specific habitat, etc. This assessment evaluates potential risks posed to aquatic biota primarily by comparing ambient PCB concentrations in surface water and sediment to media-specific and, where appropriate, site-specific criteria, standards, or critical effects concentrations (e.g., no or low observed adverse effects concentrations).

A primary output of the PCB Food Web Model is an estimation of the average potential daily dose (APDD mg PCB/kg body weight-day) from ingestion of

PCB-contaminated prey for upper trophic level organisms. This estimation is based on the following formula from EPA (1993):

$$ADD_{pot} = \sum_{K=1}^{n} (C_k * FR_k * NIR_k)$$

Where:
- ADD_{pot} = Potential average daily dose (mg PCB/kg BW-day)
- C_k = Average PCB concentration in the k^{th} food type (mg/kg)
- FR_k = Dietary fraction of intake of the k^{th} food type (range 0 to 1.0)
- NIR_k = Normalized ingestion rate of the k^{th} food type (wet weight of prey ingested per day, kg/d)
- n = Number of contaminated food types

Normalized ingestion rate is the ingestion rate normalized for body weight:

$$NIR_k = IR_k / BW$$

Where IR_k is the ingestion rate (kg/d) of the predator and BW is the body weight (kg) of the predator. As stated above, this term is expressed as wet weight, or NIR_{ww}.

For species for which incidental sediment or soil ingestion is significant, an additional term is added to the equation presented above, as shown below.

$$ADD_{pot} = \sum_{K=1}^{n} (C_k * FR_k * NIR_k) + (NIR_{dw} * PCB_{Soil} * DF_{Soil})$$

The combination of both NIR_{ww} and NIR_{dw} is required because PCB concentrations in biota serving as prey are expressed as wet weight and sediment and soil PCB concentrations are expressed as dry weight.

The site foraging factor or SFF is commonly added to the above equation (multiplied in the numerator) to account for the fact that some animals forage over a wide range. Ingestion of contaminated prey may therefore be adjusted by the portion of time foraging takes place in contaminated areas. This adjustment is most appropriate where predators with large foraging ranges are evaluated at small sites.

SFF = Site Foraging Factor

(Site area, hectares/home or foraging range, hectares) (Range = 0 to 1.0)

This ERA does not adjust the SFF and retains the SFF at 1.0, assuming that the foraging range is less than or equal to the site area. This assumption appears conservative or overly protective until one considers that nearly the entire site provides suitable habitat and food for most predators. There is no reason to believe, and there is no evidence that predators such as mink will leave the site and obtain food beyond site boundaries. All known bald eagles nests are along the Kalamazoo River and it is assumed that eagles will obtain all of their food from the Kalamazoo

River corridor. This is critical, because if a breeding pair is capable of producing fledglings, they will most likely be fed contaminated prey from the Kalamazoo River corridor. Section 5 discusses some additional evidence that supports this preliminary assumption.

Each of these input parameters, in addition to other parameters used to support the ERA (e.g., bioconcentration factors), is discussed below. Finally, for readability, the potential average daily dose (ADD_{pot}) is referred to in subsequent sections of the ERA as the APDD or average potential daily dose.

Representative Species

For assessing potential risks to ecological receptors, certain local species are selected to represent important trophic levels in aquatic and terrestrial food chains for this site. Important trophic levels for each identified food chain include primary producers (plants), primary consumers (herbivores), secondary consumers (carnivores), and top predators (carnivores at the top of a food chain). Some organisms can occupy more than one trophic position in a food web. For example, raccoons consume both plants and animals and, in some food webs, can also be considered top predators. For this assessment, forage and rough fish include both herbivorous and carnivorous species, and detritivores are included with herbivores and omnivores.

Primary Trophic Levels and Categories of Representative Organisms

Primary Producers

General categories of organisms identified as primary producers include:

- Algae
- Aquatic macrophytes
- Terrestrial macrophytes

Primary Consumers

General categories of organisms identified as being predominantly herbivorous, omnivorous, or detritivorous, include:

- Aquatic invertebrates (benthic and water column)
- Forage fish
- Rough fish
- Terrestrial invertebrates
- Small terrestrial omnivorous rodents
- Omnivorous songbirds
- Semi-aquatic herbivorous mammals

Secondary Consumers

General categories of organisms identified as being predominantly carnivorous include:

- Game fish
- Small terrestrial/semi-aquatic carnivorous mammals
- Birds of prey
- Large terrestrial carnivorous mammals

Top Predators
Secondary consumers or carnivores specifically identified as top predators for this assessment include red fox, great horned owl, bald eagle, and mink.

Local species are selected to represent general categories of organisms and important trophic levels in identified food chains. Several of these species or categories of organisms have been sampled to determine whole body PCB concentrations. Whole body (where applicable) PCB concentrations are estimated for other non-sampled species or categories of organisms. These estimates are based on species-specific BCFs or BAFs as much as possible, and on measured PCB concentrations in exposure media. For example, the PCB concentration in algae (mg/kg) is estimated by multiplying the measured surface water PCB concentration (mg/L) by an appropriately derived BCF for freshwater algae.

PCB concentrations in whole body (wet weight) or specific tissue (wet weight) are *measured* in several selected species, as summarized in Tables 4-5a and 4-5b. These species, and the associated trophic category, include:

- Terrestrial macrophytes - Based on bioaccumulation of PCBs in terrestrial plants, from data collected from onsite garden plot in 2000
- White sucker (*Catostomus commersoni*) or equivalent - forage fish
- Common carp (*Cyprinus carpio*) - rough fish
- Smallmouth bass (*Micropterus dolomieui*) - game fish
- Earthworm (*Lumbricus terrestris*) or equivalent - terrestrial invertebrate
- Deer mouse or white-footed mouse (*Peromyscus maniculatus* or *P. leucopus*) - small omnivorous terrestrial mammal
- Muskrat (*Ondatra zibethica*) - semi-aquatic herbivorous mammal
- Mink (*Mustela vison*) - terrestrial/semi-aquatic carnivorous mammal
- Bird Eggs (multiple species) - omnivorous, carnivorous, piscivorous avian receptors

PCB concentrations are **estimated** for:

- Algae and aquatic macrophytes - Based on bioconcentration of PCBs in diatoms and *Hydrilla*, respectively

- Aquatic invertebrates (benthic) - Based on bioconcentration of PCBs in scuds (*Gammarus*) and midge (*Chaoborus*) larvae determined in laboratory experiments

- Aquatic invertebrates (water column) - Based on bioconcentration of PCBs in cladocerans (*Daphnia*) and mosquito larvae (*Culex*)

- American robin (*Turdus migratorius*) - Whole body estimates based on estimated diet (using site-specific and modeled data) and diet-to-carcass BAF (alewife to herring gull) as determined by Braune and Norstrom (1989).

- PCB tissue concentrations are neither measured nor estimated for the three remaining representative top predator species: great horned owl (*Bubo virginianus*), red fox (*Vulpes fulva*), and bald eagle (*Haliaeetus leucocephalus*). This is not considered a critical data gap for three reasons:

  1. The primary purpose of determining PCB concentrations in selected organisms is to estimate potential dose through dietary exposure for consumers of contaminated prey. Top predators, by definition, are unlikely to be regularly consumed by other organisms.

  2. Data are unavailable to adequately interpret whole body or tissue PCB concentrations for these or closely related species. Contaminant body burdens are not in themselves appropriate assessment endpoints and, in general, are not useful without comparison to appropriately derived toxicity data (i.e., effects related to body burden concentrations).

  3. The primary risks associated with PCB contamination to top predators are through ingestion of PCB-contaminated prey, and available toxicity data primarily relate toxic effects to dietary dose rather than to PCB concentrations in whole body or specific tissue type.

For these reasons, estimations of the average potential daily dose (APDD) from ingestion of contaminated prey are used to assess potential PCB-related risks for the great horned owl, red fox, and bald eagle.

Input Parameters and Assumptions

The following subsections show the model input parameters, as well as assumptions made for each. Appendix C-1 includes all input parameters, thresholds or criteria, and associated assumptions for all media and receptors. Appendix C-2 shows the calculations for PCB doses, hazard quotients (HQs), and PRGs for terrestrial and semi-aquatic receptors. Appendix C-2 consists of two parts. C-2-A is a spreadsheet used to calculate doses, HQs, and PRGs for terrestrial receptors, and C-2-B is a similar spreadsheet for semi-aquatic receptors.

PCB Concentration

Where data quantity allow, PCB concentrations are based on the U95 concentration of PCBs in abiotic media (surface water, streambed and floodplain sediment, and surface soil) of concern. These values are based on specific terrestrial and aquatic biota sampling areas (TBSAs and ABSAs), as described in the *Biota Sampling Plan* (CDM 1993). U95 values are also used to describe PCB concentrations in biological tissues if sufficient data have been collected to allow for U95 calculations. Where data are more limited (e.g., terrestrial biota), maximum detected values are used for the reasons discussed previously. Values are in mg PCB/L for surface water and mg PCB/kg (dry weight) for sediments, surface soil (from TBSAs), and biological tissue.

PCB concentrations in surface water (mg/L), streambed and floodplain sediment (mg/kg), and surface soil (mg/kg) are based on measured values. PCB concentrations in biological tissue (mg/kg, wet weight) are estimated for aquatic organisms considered representative of lower trophic levels. These organisms include algae, aquatic macrophytes, and aquatic (benthic and water column) macroinvertebrates. In addition, PCB concentrations are estimated for birds, represented by American robin, from calculated PCB concentration in robin diet, using literature-based diet to whole body (carcass) data for birds. PCB concentrations for earthworms (depurated), all fish species, muskrat, mink, and mice are based on the ABSA- or TBSA-specific maximum measured whole body (and liver for mink and muskrat) PCB concentration for these organisms. Terrestrial plant PCB concentrations are based on measured garden plot data for several crop species from ABSA 8, collected in 2000. For species likely to eat fruits or berries (e.g., robin and fox), the BAF determined for tomatoes at this location was used to estimate PCB concentrations in fruits and berries. PCB concentrations were neither measured nor estimated in the remaining three species (great horned owl, red fox, bald eagle) for the reasons cited previously.

Exposure Media

Exposure media represent the primary media to which specific receptors or categories of receptors may be exposed. These media include surface water, streambed and floodplain sediment, and surface soil. Streambed sediments are bottom sediments covered with surface water. Floodplain sediments are those sediments deposited behind former impoundments, and may or may not be dry depending on specific location and season. Floodplain sediments that are inundated for several months each year are best viewed as streambed sediments for the purposes of food chain modeling and derivation of preliminary remedial goals (PRGs). Floodplain sediments that are never inundated or only rarely wet should be viewed as surface soils. Media identified as *surface soils* specifically refer to those soils collected within TBSAs. TBSA soil samples may include samples taken from perennially dry areas representing true terrestrial exposures as well as samples taken from seasonally inundated areas. The latter are more appropriately considered floodplain sediments, and are more closely associated with aquatic exposures. Surface soils are also assumed to best describe those solid media found in upland areas, including areas associated with elevated landfills. Finally, floodplain sediments for ABSA 11 (Ottawa and Potawamie Marshes)

are identified as wetland/marsh sediments that differ from sediments associated with the former impoundments.

Bioconcentration or Bioaccumulation Factor

BCFs/BAFs (Aquatic)
BCFs are based on the ratio of tissue contaminant concentrations in species of concern (mg/kg) to contaminant concentrations in surface water (mg/L). Bioconcentration considers only direct uptake from water, and does not include uptake from food. In general, BCFs are used for aquatic plants, aquatic invertebrates, and fish, and are based on laboratory tests in which sediments and contaminated prey are absent. Some BCFs presented in Appendix C-1 are derived from literature-based values and are applicable where specific biota such as algae, aquatic macrophytes, and aquatic invertebrates were not sampled. Laboratory-derived BCFs may not reflect bioconcentration potential under field (i.e., natural) conditions. For this study, the uptake of PCBs by algae, aquatic macrophytes, and aquatic invertebrates is estimated from appropriately-derived (i.e., following EPA guidelines) geometric mean BCFs in the literature, while BCFs (actually BAFs) for fish are calculated from site-specific measured U95 PCB concentrations in surface water and fish. There is greater confidence in the calculated BAFs for fish compared to BCFs for algae, aquatic macrophytes, and aquatic invertebrates. Confidence in the field or site-specific BCFs is increased because these data reflect uptake from all sources, not just water. Confidence in these same values is decreased to some degree because the fish and surface water data were not collected at exactly the same times and locations. These relationships are, however, considered useable because the surface water and fish data were collected within approximately the same time period and are ABSA-specific.

BAFs (Terrestrial)
BAFs are similar to BCFs except that they reflect uptake from both food and water. The uptake of contaminants by fish and other aquatic organisms exposed to contaminated surface water, sediment, and prey in the field is best described using BAFs rather than BCFs.

BAFs can also be used to describe the soil-to-plant transfer of contaminants in terrestrial systems. For this assessment, BAFs for terrestrial macrophytes are based on one of two values.

- For diets composed of multiple types of plant tissues (e.g., roots, stems, leaves, fruits, and seeds, estimated plant PCB concentrations are based on the upper 95th confidence limit of the arithmetic mean measured co-located soil and plant PCB concentrations from a garden plot in ABSA 8 or

- For diets composed primarily of fruits or berries, estimated plant PCB concentrations are based on measured co-located soil and tomato PCB concentrations from a garden plot in ABSA 8.

These data were collected in part in response to KRSG comments (September 11, 2000 letter) on the lack of site-specific soil-to-plant bioaccumulation factors. These data were obtained in 2000, and are based on eight crop species. These soil and plant PCB concentrations, along with calculated BAFs for co-located samples, are presented below in Table 4-6. This ERA uses the site-specific U95 BAF of 0.037 to estimate general plant uptake and PCB doses for herbivorous receptors likely to consume a variety of plant tissues such as leaves, stems, and seeds. Calculated PCB doses for herbivorous or omnivorous receptors expected to consume primarily fruits (e.g., robin) are based on the soil to tomato BAF of <0.0008 (set to 0.0008). It is recognized that these BAFs may overestimate or underestimate PCB uptake for terrestrial plant species because of uncertainties related to sample size and PCB uptake in plant species and tissue types (e.g., seeds) likely to be consumed by certain representative herbivorous or omnivorous receptors.

To provide other lines of evidence regarding plant uptake of PCBs, Table 4-7 presents other literature-based values for PCB transfer from surface soil to terrestrial plants. The soil-to-plant transfer factors or BAFs presented on Table 4-7 are ranked from lowest to highest. The site-specific BAF of approximately 0.04, from the garden plot data, is also included on this table and is identified in bold type. It can be seen that the selected site-specific soil-to-plant BAF of 0.04 falls approximately at the mid-point of the ranked literature-based data. These literature-based data include experimental and modeled BAFs, and are believed to encompass the range of values that may be observed in the field with a variety of plant species and tissue types. It is noted that species and plant tissue types (e.g., seeds) that are likely to be consumed by herbivorous or omnivorous consumers such as deer mice are not included in this list of literature-based plant BAFs. Although this is an area of uncertainty, the garden plot data and resulting BAFs (0.037 and 0.0008) are considered adequately representative of soil-to-plant PCB transfer at this site.

The results of some studies presented in Table 4-7 indicate that certain terrestrial plants can accumulate PCBs from soil to a concentration greater than the original soil concentration (i.e., BAF>1). Trapp, et al. (1990) presents the results of two experiments in which the average plant PCB concentration was approximately 1.3 times that of the soil in which the plant was grown. Pal, et al. (1980) described biomagnification factors (BMFs) for several plant species. As expected, most terrestrial species accumulated PCBs from the soil at a BAF (or BMF) of less than 1.0. However, included in this list of BMFs for several plant species are two results that support a higher BAF for some species. Carrots, for example, accumulated PCBs from the soil at a factor of about 0.25, while weeds exposed in the same study accumulated up to a factor of 0.96 times the soil concentration (i.e., BAF = 0.96). Weeds exposed in a study focused on sugarbeet accumulation of PCBs took up PCBs from the soil at a factor of 0.80 (BAF = 0.80).

Much higher BAFs are described by Pal, et al. (1980) for aquatic and riparian plants that occur in wet soils or soils that are frequently flooded.

BAFs are also calculated from measured PCB concentrations for most of the remaining aquatic, semi-aquatic, and terrestrial species. In cases where more than one media type is identified as a potential source of PCB contamination, BAFs are based on the primary exposure media. For example, mink feed on a wide variety of aquatic, semi-aquatic, and terrestrial animals. PCB contamination in surface water, streambed and floodplain sediment, and surface soil can all contribute to PCB accumulation in mink through ingestion. For this reason, it is inappropriate to calculate BAFs or PRGs based on multiple, often uncertain exposure scenarios. Food chain modeling for mink is limited in this ERA to calculation of doses used to derive hazard quotients. Calculated aquatic (surface water) and terrestrial (surface soil) BAFs are based on TBSA/ABSA-specific PCB concentrations measured in abiotic exposure media and biota (Table 4-8), where these data are available. In addition, Table 4-8 presents BSAFs for ABSAs where streambed sediment and fish were collected over approximately the same time period. BSAFs reflect the potential transfer of a contaminant in sediment to biological tissues. The confidence in the ABSA-specific BSAFs is increased by the relatively large amount of fish and sediment data collected over approximately the same time period from the same ABSA. Contributing to decreased confidence in these BSAFs is the fact that the fish and sediment data were not collected at exactly the same location and time. The latter is not considered a critical data gap because of the mobility of fish and the variability in sediment PCB concentrations within an ABSA.

Diet-to-Bird BMF

Site-specific data are lacking for PCB concentrations in whole body birds. Whole body bird PCB concentrations must therefore be estimated from available site-specific data (e.g., PCB concentrations in worms and plants) and literature-based data (e.g., biological multiplication factor (BMF) that relates PCBs in diet to whole body burden). Literature-based BMFs have been reviewed for use in this ERA for estimating total PCB concentrations in whole body birds from bird diets. The selection of the most appropriate BMF is important because the consumption of whole body birds contributes to modeled total PCB dietary doses (and risks) for great horned owl, red fox, bald eagle, and mink.

The diet-to-bird BMF selected for food chain modeling in this ERA is 93, taken from Braune and Norstrom (1989). This BMF is based on PCB-contaminated fish (alewife) consumed by herring gulls. The BMF (93) from Braun and Norstrom was also used for total PCBs in the Great Lakes Initiative (rounded to 90) for estimating risk to bald eagle (USEPA 1995b). This peer-reviewed EPA document is used for regulatory purposes. The BMF of 93 is also consistent with a caged juvenile herring gull feeding study that resulted in a diet-to-bird BMF of 97 (quantified as A1254 and described as "apparent PCBs", Anderson and Hickey 1976).

Additional supporting information is used to confirm the consistency of the Braun and Norstrom study with other similar studies. This included a comparison of diet-to-egg BMFs. Diet-to-egg BMFs are not used directly in this ERA but data from two separate studies are compared here to provide additional support for using the Braun and Norstrom BMF data.

Lipid-normalized diet-to-egg BMFs for individual PCB congeners in the Braun and Norstrom study are consistent with (and actually lower than) the lipid-normalized fish-to-egg geometric mean congener BMFs calculated by Blankenship and Giesy (2002) from multiple studies. Lipid-normalization is based on the following lipid contents reported by Braun and Norstrom (1989): herring gull whole body - 10.3 percent, gull egg - 7.7 percent, and alewife - 2.8 percent.

Congener-specific lipid normalized diet-to-egg BMFs are presented below for both the Braun and Norstrom (B&N) study and the geometric means calculated by Blankenship and Giesy (B&G). The Braun and Norstrom (B&N) data presented below include additional congener data not included in the original paper (1989) but subsequently reported by Hoffman, Rice, and Kubiak (1996).

PCB Congener	77	101	105	110	118	126	138	153	169
B&G	0.89	4.52	7.95	5.4	26.15	29.74	27.74	32.57	31.25
B&N + H,R&K	0.7	2.9	7.3	2.5	11.3	10.5	17	17.3	16.7

The total PCB lipid-normalized diet-to-egg BMF from the Braun and Norstrom study is 11.5. This is comparable to the total PCB geometric mean lipid-normalized diet-to-egg BMF of 18.1 (range 10.4-36.8) reported by Koslowski, et al. 1994 for Lake Erie gulls--one of the studies relied on by Blankenship and Giesy (2002). Blankenship and Giesy (2002) did not, however, report total PCBs.

As discussed above, the Braun and Norstrom BMFs are supported by the results of several studies. However, substantially lower diet-to-bird BMFs of 10 or less for total PCBs have also been reported in the literature. This leads to uncertainty with the diet-to-bird BMF expected in the field. The more conservative (higher) BMF determined by Braun and Norstrom is selected for this ERA because regulatory guidance recommends using a conservative or more protective approach where uncertainty exists.

Finally, the value assigned to the diet-to-bird BMF affects food chain modeling for only the great horned owl, red fox, bald eagle, and mink, in decreasing order of importance. The order of importance is based on the estimated dietary fraction comprised of birds for each of these receptors. The estimated dietary fraction of birds is 47 percent for great horned owl, 19 percent for red fox, 17 percent for bald eagle, and 5 percent for mink. The diet-to-bird BMF influences to a small degree the risk estimates (i.e., hazard quotients) for mink, but does not affect the PRGs established for protection of mink, which are not based on food chain modeling.

Home Range

An animal's home range can greatly affect its degree of exposure. For example, animals with home ranges entirely within a contaminated area will have greater exposure potential than animals with home ranges that substantially exceed the area of a contaminated site. This assumption may not always hold true, however, because home range values are often only estimates of the average area used by a particular species. It is not unreasonable to assume that an animal with a large home range will, at times, remain within a smaller area if that area provides adequate food and cover. In addition, models that estimate dietary exposures, including this model, are very sensitive to variability in home range estimates. Average home ranges for adult animals are presented in the model.

Site Foraging Frequency

Standard practice in assessing dietary exposures for wildlife includes the derivation of site foraging frequency (SFF). This term is used to describe the ratio of the site area to the average home range for the species of concern. As commonly used, SFF values range from 0 to 1.0. It is apparent that animals with large home ranges are less likely to be significantly exposed to site-related contamination than animals that live entirely within site boundaries. However, as stated above, the use of home ranges for estimating exposure likelihood has certain critical limitations. First, home range estimates are based on overall use, yet certain individuals or populations may use smaller areas for foraging and cover if conditions are suitable. Also, dietary exposure models are extremely sensitive to variability in the input parameter identified here as SFF. It is not uncommon for dietary exposure models to predict zero or nearly no risk for species associated with highly contaminated sites solely because their average home range is very large. The API/PC/KR area is large, and areas of PCB contamination are not evenly distributed in size or location. Thus, accurately correlating home range to site area is difficult at this site for species with large home ranges. However, this ERA focuses on those species that would primarily spend all or most of their time within the Kalamazoo River corridor.

Finally, the methods for determining home ranges are not intended to support the specific needs of ecological risk assessment. Home range sizes, which are presented in Appendix C, are often determined by locating nests, dens, or spawning areas for species of concern and then recording the locations of individual organisms observed in the area of the nest or den. Locations of individual organisms observed are then plotted on a map and connected by lines forming a polygon, with the nest or den located within the polygon.

The area of the resulting polygon is considered to be a home range. This method does not consider frequency and size of foraging areas within the estimated home range, and therefore may be inappropriate for ecological risk assessment use. For the reasons cited, this assessment sets the SFF to 1.0 for all species for which dietary exposure is calculated. Although this adds conservatism to the model, it is considered prudent to prevent gross under-estimations of potential risks for some ecological receptors.

Dietary Fraction

Dietary fraction is an estimate of the fraction of total diet contributed by each prey type. For this study, estimates of dietary fraction are based on values reported in the literature. Where more than one literature source of dietary information is available, estimates are based on the average of all relevant literature sources (primarily EPA 1993) or the values most relevant to Western Michigan. The fraction of soil or sediment incidentally ingested is also included if such ingestion is deemed appropriate. For example, muskrat are assumed to incidentally ingest a substantial amount of sediment while feeding and grooming, while bald eagles feeding in a riverine environment on fish probably ingest little or no sediment.

Average Ingestion Rate

Average ingestion rates (kg/d) are determined for species of concern from values in the literature. Most data are taken from EPA's *Exposure Factor Handbook, Volume I* (1993). Ingestion rates are presented as both wet weight and dry weight—the latter is used where ingestion of sediment or soil is significant. Sediment and soil PCB concentrations are expressed as mg/kg dry weight, while plant and animal dietary items are expressed as mg/kg wet weight.

Average Body Weight

Average body weights (kg) for representative adult organisms are based on values presented in literature sources. Where more than one source was consulted, the value used is based on the average of all species-specific adult body weights presented. In some cases, average body weights can be substantially different for males and females of the same species. Where this is the case, values used are based on the average of values reported for adult males and females.

Model Output

As stated above, the primary model output is an estimate of the average potential daily dose (APDD, mg PCB/kg BW-d) for upper trophic level organisms from ingestion of contaminated prey. This value is not determined for lower trophic level organisms (e.g., algae, macroinvertebrates, earthworm, forage fish) or game and rough fish because either it is not applicable (e.g., algae) or input parameters (e.g., ingestion rates) are generally unknown or associated with a high degree of uncertainty. APDD values may over- or underestimate actual PCB doses because of site-specific diet or foraging habits. Also, actual PCB doses probably vary seasonally and spatially.

For organisms for which APDD is not calculated, risk estimations are based on comparisons of exposure point concentrations of PCBs (e.g., PCB concentration is surface water) to NOAECs, LOAECs, criteria, or recommended limits.

Average Potential Daily Dose, APDD, (mg PCB/kg BW-d) is calculated from the equation described previously, and serves as the primary output of the PCB Food Web Model. This value is used to estimate potential risk to upper trophic level

organisms from ingestion of contaminated prey by comparison with critical dietary concentrations.

Toxicity Assessment
The potential toxicity of PCBs to representative organisms is evaluated by comparing measured or estimated PCB concentrations in abiotic media or prey to

- appropriate media-specific criteria (e.g., AWQC),
- safe levels not associated with adverse effects (e.g., NOAECs or EC_{10}/ED_{10}), or
- species-specific concentrations at which adverse effects begin to be observed (e.g., LOAECs or EC_{25}/ED_{25}).

Although considered part of the food web model as a preliminary evaluation, these data are further discussed in the Effects Assessment portion of the ERA. The effects assessment also discusses other effects data used in the Risk Characterization phase of the ERA, including site-specific values with which overall risks to ecological receptors are evaluated.

No Observed Adverse Effects Concentration (NOAEC)
NOAECs are obtained from the literature for species of concern or for closely related species that are expected to exhibit toxicologically similar responses to PCB exposures. Species-specific NOAECs are compared to measured or estimated PCB concentrations from similar routes of exposure (e.g., direct contact or ingestion of food items) for selected species. Specific NOAECs selected for this study include the highest concentrations associated with no adverse effect from toxicity tests conducted with species of concern. Also consulted are primary data sources referenced in EPA contaminant-specific criteria documents (aquatic organisms) and FWS contaminant hazard review documents (terrestrial organisms). NOAECs are not associated with adverse effects; therefore, PCB concentrations at or near the relevant NOAECs are assumed to be associated with no risk. NOAECs are commonly estimated by (LOAEC/10). Based on the comparison of two studies performed with field-contaminated fish, Giesy, et al. (1994) recommended the use of LOAEC/3 for estimating NOAECs for mink exposed to PCBs through diet. A review of available data for certain species of birds and mammals supports the recommendation of Giesy, et al. This ERA uses LOAEC/3 to estimate NOAEC for mouse and muskrat and uses NOAEC * 3 to estimate LOAEC for great horned owl. The phrase No Observed Adverse Effects Level or NOAEL is used when exposure is expressed as dose (i.e., mg/kg-d). A different (ED_x or EC_x) approach, discussed below, is used to derive the no effect and low effect toxicity reference values (TRVs) for mink and non-raptor birds.

Lowest Observed Adverse Effects Concentration (LOAEC)
LOAECs are also obtained from the literature for species of concern or for closely related species that are expected to exhibit toxicologically similar responses to PCB

exposures. Similar to NOAECs, species-specific LOAECs are compared to measured or estimated PCB concentrations from similar routes of exposure (e.g., direct contact or ingestion of food items) for selected species. LOAECs are by definition associated with adverse effects; therefore, PCB concentrations at or near the relevant LOAECs are associated with some, possibly unacceptable risk. LOAECs based on dose are termed Lowest Observed Adverse Effects Levels or LOAELs. As mentioned above, a different approach is used to derive the no and low effect TRVs for mink, American robin and bald eagle. Owl-specific toxicity data are used to assess risks to great horned owls. A summary of this approach follows.

Effect Concentration (EC_x) / Effect Dose (ED_x)

It can be difficult to determine the most appropriate no effect and low effect TRVs for mink and non-raptor birds exposed to PCBs based on reported NOAELs and LOAELs. Such difficulties arise because of significant differences in the methodologies and designs of studies in which mink and non-raptor birds are fed PCB-contaminated food. Important differences include test endpoints, chemical form of PCBs fed, test duration, and potential confounding effects of other contaminants present in food items. These differences result in varying degrees of confidence in reported or calculated doses defined as NOAELs. For this reason, there are often disagreements on the appropriateness of any given NOAEL or LOAEL defined as a preferred TRV. As an alternative to selecting a single NOAEL or LOAEL, this ERA uses a more detailed analysis of toxicity data to derive the no effect and low effect TRVs for mink and non-raptor birds. The approach is introduced below for mink and birds, with a more detailed discussion of these TRVs in Section 4.2, Effects Assessment.

MINK - The no and low effect TRVs for mink are based on a detailed analysis of the literature on the effects of PCBs on mink. The TRVs for mink, which form the basis of the surface water and sediment preliminary remedial goals (PRGs) for this site, are discussed in detail in Section 4.2. The calculated dietary PCB low effect TRV for mink is 0.6 mg/kg wet weight (diet). The estimated no effect TRV is 0.5 mg/kg wet weight (diet) for mink. In addition to the discussion of mink TRVs in Section 4.2, Appendix D provides a complete and detailed discussion of the method used to derive these TRVs.

BIRDS - The no and low effect TRVs for birds (i.e., American robin and bald eagle) are based on a detailed analysis of the effects of PCBs on chicken, one of the best-studied and most sensitive avian receptors of the few species investigated to date. The TRVs for non-raptor birds are discussed in detail in Section 4.2, Effects Assessment. The calculated low effect TRV for birds is 0.5 mg/kg_{BW}-d, based on Aroclor 1248, the predominant Aroclor detected in earthworms in the Kalamazoo River floodplain. The calculated no effect TRV for birds is 0.4 mg/kg_{BW}-d, also based on Aroclor 1248. Appendix D presents a detailed summary of the ED_x/EC_x method used to derive TRVs for birds other than great horned owl, and Section 4.2 presents a more detailed analysis of the final TRVs selected for these birds.

Criteria or Recommended Limits
In some cases, criteria (e.g., AWQC) or maximum allowable limits (e.g., those recommended for the protection of sensitive birds or mammals) have been established for species or other taxa of concern. Where such values are available, they are presented in the food web model for comparison to measured or estimated PCB concentrations determined in this study. Criteria and limits presented in Appendix C are not site-specific but are instead based on general toxicological data. The comparisons between toxicological data from the literature and exposure data for this site are used to evaluate reasonable maximum exposures for the API/PC/KR site, based on U95 PCB concentrations in abiotic and most biological media.

A comparison of arithmetic average PCB exposure data to toxicological data may also be useful, but is considered less appropriate for a large and diverse site like the API/PC/KR. The API/PC/KR site is associated with highly variable abiotic PCB concentrations from one area to another, and average measured concentrations of PCBs are not likely to represent the true average or especially the reasonable worst-case exposure. U95 and, in cases where sample size is small, maximum ABSA- and/or TBSA-specific exposure concentrations are therefore preferred for evaluating potential effects in ecological receptors.

Preliminary Remedial Goals (PRGs)
This ERA develops a range (i.e., no effect to low effect) of site-specific PRGs to be considered as remedial goals associated with the protection of key receptors or habitat types. Where data allow, these site-specific PRGs are based on measured PCB concentrations in exposure media and food items as well as site-specific bioaccumulation in sampled biota. The equations used to calculate terrestrial and aquatic PRGs are presented below. PRGs are presented in the risk characterization phase of the ERA, and the derivation of receptor-specific PRGs is presented in Appendix C-2. The first example is for terrestrial receptors that are assumed to ingest soil along with prey.

Terrestrial SED/SOIL PRG =

$$\textit{(No Effect or Low Effect TRV / SUM (NIRww * } BAF_{Prey}\textit{1...x *}^{DFPrey}\textit{1...x) + (}NIR_{dw} * DF_{Soil}\textit{))}$$

Where:

No Effect or Low Effect TRV = Species-specific dose (mg PCB/kg BW per day)

NIR_{ww} = normalized daily ingestion rate (IR / BW), mg/kg-d, wet weight

BAF_{Prey} = bioaccumulation factor for PCBs in prey item

DF_{Prey} = dietary fraction of prey ingested

NIR_{dw} = normalized daily ingestion rate (IR / BW), mg/kg, dry weight

DF_{Soil} = dietary fraction of soil/sediment ingested

PRGs for mink exposed to aquatic and semi-aquatic (seasonally inundated) sediments are based on surface water PCB thresholds derived to protect fish tissue from accumulating critical levels of PCBs. These PRGs also consider the site-specific relationships between PCBs in surface water and sediments. The general equation for deriving aquatic PRGs is presented below. Two different ways of viewing this derivation are presented.

Aquatic SED PRG for Mink Protection

= SW threshold * SW-to-SED Partition Factor

or

Aquatic SED PRG for Mink Protection

= Fish Tissue Threshold/BSAF

Where:

BSAF = biota sediment accumulation factor

The fish tissue threshold is based on the surface water threshold and site-specific bioaccumulation of PCBs into fish tissue. The surface water to sediment partition factor is the mean site-specific value for co-located surface water and sediment PCB concentrations. These two equations are therefore mathematically related and are not different. Section 4.2.1 shows these PRG derivations in greater detail.

Site-specific effects data are presented in Section 4.2, Ecological Effects Assessment, and are further discussed in Section 5, Risk Characterization, where risk estimates and proposed cleanup goals or PRGs are presented. An interpretation of the output of the food web model Appendices C-1 and C-2 is presented in the Risk Characterization section of the ERA. The Risk Characterization section discusses the results of the food web model and integrates exposure and effects data to estimate risks to ecological receptors of the API/PC/KR. Effects assessment follows an analysis of uncertainties associated with exposure analysis and the food web model.

4.1.6 Uncertainty Evaluation – Exposure Assessment

Sources of uncertainty in the exposure assessment include the values used to represent the magnitude and distribution of media-specific contamination. Obviously, all media cannot be sampled at all locations, and data interpolation and/or extrapolation are necessary. It is expected that the samples collected have been appropriately analyzed to adequately describe the nature and extent of PCB contamination at the API/PC/KR site. Uncertainty in this assessment is decreased by

the biological sampling specifically designed to support food web modeling and to support descriptions of the magnitude and distribution of PCB contamination at the API/PC/KR site. Because ABSA and TBSA-specific sampling was relatively complete for abiotic media, the use of U95 concentrations of PCBs in SW, SED, FP SED, SS, and most biota minimize the chance that risk estimations based on the selected exposure concentrations have been greatly under- or over-estimated.

Another major source of potential uncertainty in the ERA is the food web model. All models, including simplified models such as the one described herein, are associated with uncertainty. In general, more complex bioenergetic-type models have greater potential to accurately estimate contaminant transfer between environmental compartments but also have greater potential to introduce unacceptable levels of uncertainty unless critical information on site-specific input parameters are available.

For example, aquatic food web models based on bioenergetics have been established that calculate biomagnification factors (BMFs) for organic contaminants from exposure media through all major trophic levels to top predators. These models often require the use and evaluation of input parameters that are currently unknown, such as contaminant depuration rates for a particular species. Values for other species or even other chemicals are sometimes used to represent the required input parameter.

Models may also be sensitive to slight differences in input parameter values, and results can, therefore, be highly uncertain. The uncertainty in resulting BMF estimations for higher trophic level organisms are also magnified because the model is based on addition and multiplication of values from lower trophic levels. For these reasons, complex computer-based food chain models are not considered appropriate for this assessment.

Although every caution was taken in this assessment to limit uncertainty as much as possible, simple models can also be associated with uncertainty. Where potential levels of uncertainty could adversely affect the results of the assessment, conservative approaches were taken that may result in over-protection of some local species. For example, many simple food chain models commonly predict, largely as a result of home range estimates, little or no risk to top predators from ingestion of contaminated prey. The SFF calculated from large home range estimates can therefore "drive" the model output (i.e., the APDD) for certain potentially important species. As discussed above, the foraging behavior of individual organisms and even populations are sufficiently unknown to warrant a more conservative or protective approach. To err on the side of over-protection is considered prudent and, in fact, follows regulatory guidance.

The most likely causes of uncertainty in this assessment are the variability of values associated with certain input parameters, especially values used to describe the distribution of PCB contamination in various media and biota. There is greater uncertainty in PCB concentrations estimated for certain prey items. For example, PCB

concentrations are estimated (using a literature-based BMF) for whole body birds that serve as prey for certain representative receptors (great horned owl, red fox, bald eagle, and mink). These estimated whole body PCB concentrations in birds are based on modeled PCB concentrations for robin using the literature-based BMF and site-specific data for plants and worms comprising robin diet. PCB concentrations in robin diet include a significant exposure via consumption of earthworms. Birds that consume mostly seeds or fruits are likely to have lower PCB exposures than those that eat mostly earthworms. Also, the selected diet-to-bird BMF (93, from Braun and Norstrom 1989) exceeds the diet-to-bird BMF determined in some other studies. The combined impacts of using a vermivore to represent songbirds and using a high diet-to-bird BMF probably overestimates risks to predators of songbirds. On the other hand, risks may be underestimated for predators of piscivorous birds such as mergansers, herons, and kingfishers.

Using U95 values for the larger abiotic and biological media data set and maximum values for the smaller biological data sets is expected to limit uncertainty and risk under-estimation to an acceptable degree. Literature values for BCFs and, to a lesser degree dietary fractions, are also critical with regard to potential for uncertainty due to uncertainties associated with laboratory to field extrapolations. There is more confidence in values used to represent species-specific ingestion rates and body weights because, in most cases, there is reasonable concurrence by investigators. Finally, NOAECs, LOAECs, EC_{10}, ED_{10}, EC_{25}, ED_{25}, criteria, and recommended limits are often based on literature values derived under controlled conditions that may not be fully relevant to natural field conditions. Also, certain criteria or recommended limits are usually intended to protect large and diverse groups of organisms (i.e., aquatic life, mammals, etc.). These values may therefore be over- or under-protective of certain local species and/or populations.

Uncertainty in this assessment regarding field-generated data is likely to be limited mostly to uncertainties in the representativeness of biological samples. Such samples are expected to be highly variable even within a species because of differences in individual behavior and activities. Even these factors are expected to vary from season to season and from one location to another. These types of uncertainties provide one basis for using maximum detected concentrations of PCBs in biological tissues for risk estimations. It is therefore more unlikely that this assessment underestimates risk because conservative approaches such as these are used where appropriate, and any uncertainties are probably biased towards over-protection.

4.2 Ecological Effects Assessment

Effects Assessment includes an evaluation of data sources and data types, and presents media-specific and stressor-specific ecological effects concentrations for PCBs, the primary chemical stressors identified at the API/PC/KR. These data serve as major components of stressor-response profiles, which describe the relationship between ecological stressors and effects. Certain types of effects data, such as NOAELs/No Effect Levels and LOAELs/Low Effect Levels, form the basis for the PRGs developed to protect key receptors representative of particular exposure scenarios and receptor groups.

4.2.1 Evaluation of Effects Data

This section of the ERA describes and provides support for the sources and types of effects data (e.g., toxicity data) selected for use in the ERA. Data sources and types are described on a media-specific basis. Selected measurement endpoints or effects data are based on relevance to the API/PC/KR site, and site-related stressors and receptors are considered in this selection. These data are directly applicable to assessment endpoints and remedial action objectives determined for the API/PC/KR site which include:

1. The preservation of the survival, growth, and reproduction of wildlife

2. The establishment and maintenance of a healthy and diverse aquatic ecosystem in and adjacent to the API/PC/KR site

3. Reductions in PCB concentrations through removal and destruction of contaminated media

4. Reductions in PCB concentrations in fish and wildlife such that human consumption restrictions can be lifted

Some effects data are more relevant and useful than others. For example, effects data are unavailable for certain receptors or receptor groups associated with the API/PC/KR. In these cases, the effects assessment is based on more general effects data available in the literature. Finally, site-specific data, such as bioconcentration and bioaccumulation factors determined by recent sampling and analysis of media and biota, are used to support estimations of risks for ecological receptors. The effects assessment provides multiple lines of evidence using numerous data sources to evaluate risks. This approach is especially important where relevant site-specific data are limited. The availability of effects data is media specific, and relevant data sources for each media of concern are presented below.

Effects Data Sources (Surface Water)

Acceptable and relevant effects data for PCBs in surface water are generally available. More general (i.e., not site specific) surface water toxicity data used in this ERA are from the EPA *Ambient Water Quality Criteria (AWQC) document for Polychlorinated*

Biphenyls (EPA 1980) and Polychlorinated Biphenyl Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review (Eisler 1986). The chronic AWQC derived by EPA is based on protection of mink (the most sensitive wildlife species tested) and considers fish ingestion by mink.

Site-specific surface water total PCB concentrations are also derived to protect mink, under the assumption that protection of mink results in protection of all other less sensitive receptors. These protective values are based on limiting total PCBs in mink diet to levels associated with no effects and low levels of adverse effects. These two values, No Effect and Low Effect dietary toxicity reference values (TRVs), form the basis for the surface water total PCB thresholds designed to protect mink at this site.

As discussed in Section 4.1, it can be difficult to determine the most appropriate no effect and low effect TRVs for mink exposed to PCBs based on reported NOAELs and LOAELs. This ERA therefore uses a different (EC_x) approach to derive the no effect and low effect TRVs for mink. The no and low effect TRVs for mink are based on a detailed analysis of the effects of PCBs on mink. The TRVs are derived from exposure-response curves by interpolation of the effective dietary concentration (EC_x) to female mink that corresponds to specific relative responses (calculated as the treatment response divided by the control response). The low effect level is defined as 0.75 of the control response for a toxicological endpoint (EC_{25}, which represents a 25% decrease in response) and the no effect level is equal to 0.90 of the control response (EC_{10}, which represents a 10% decrease in response). Appendix D provides a more detailed analysis of this approach.

The calculated dietary PCB low effect TRV for mink is 0.6 mg/kg wet weight (diet) based on the effects of Aroclor 1254 on the number of live kits per mated female and kit body weight, adjusted for continuous exposure through two breeding seasons or generations; and the no effect TRV is 0.5 mg/kg based on the effects of Aroclor 1254 on the number of live kits per mated female, adjusted for continuous exposure through two breedings seasons or generations.

The 0.5 and 0.6 mg PCB/kg dietary thresholds for mink are used to calculate a threshold surface water concentration that is protective of mink that consume PCB-contaminated fish. The mean of the average BAF for carp, smallmouth bass, and sucker is used to estimate PCB uptake in fish. This mean BAF is 305,000, as presented on Table 4-8. This BAF and the dietary No Effect TRV 0.5 mg/kg is used to calculate the surface water (SW) threshold associated with no adverse effects.

The SW threshold presented below is based on the average water-to-fish BAF (mean of the mean BAF for all three species) and the assumption that mink diet is comprised of 100 percent fish, with each of the three fish species representing one third of the diet. This conservative approach is based on the need to maintain PCB concentrations in the primary food of mink (fish) at levels that are protective of mink.

No Effect SW threshold

$$= \frac{0.5 \text{ mg PCB/kg fresh weight diet}}{305{,}000}$$

$$= 0.0000016 \text{ mg PCB/L water}$$

$$= \mathbf{0.0016\ \mu g\ PCB/L\ water}$$

The surface water threshold calculated to prevent whole body fish from containing more than 0.5 mg PCB/kg wet weight is 0.0016 µg/L.

Similarly, a Low Effect SW threshold is calculated using the same mean BAF and the Low Effect dietary threshold of 0.6 mg/kg.

Low Effect SW threshold

$$= \frac{0.6 \text{ mg PCB/kg fresh weight diet}}{305{,}000}$$

$$= 0.00000197 \text{ mg PCB/L water}$$

$$= \mathbf{0.00197\ \mu g\ PCB/L\ water}$$

The surface water threshold calculated to prevent whole body fish from containing more than 0.6 mg PCB/kg wet weight is 0.00197 µg/L.

Effects Data Sources (Sediment)

Universally accepted biological effects concentrations for most sediment contaminants have not been developed for ecological receptors. In general, the most useful data on potential sediment toxicity is obtained from site-specific studies using site sediments and resident or representative test species.

Site-specific sediment toxicity data are unavailable for this ERA. The evaluation of the potential toxicity associated with PCB contamination of onsite streambed sediments is based on the comparison of PCB concentrations in API/PC/KR streambed sediments to various relevant data. These include background concentrations, EPA-recommended and site-specific sediment concentrations based on the equilibrium partitioning (EP) approach (EPA 1988b) using both literature-based and measured (site-specific) input parameters (e.g., sediment/water partition coefficients or K_ds), and other relevant data from sources such as Long and Morgan (1991) and Persaud, et al. (1993). Databases such as that of Long and Morgan (1991) have been established that describe the co-occurrence of chemical contaminants and apparent biological effects, and others (e.g., Persaud, et al. 1993) include interim criteria for contaminants in sediment. Although the data presented in these more general (i.e., non-site-specific) databases are associated with certain limitations and uncertainties, they can contribute useful information to the overall evaluation of potential sediment toxicity

using a weight-of -evidence approach. Such an approach is used in the risk characterization phase of this ERA. There, sediment toxicity data are supplemented with comparisons between onsite PCB concentrations in API/PC/KR sediments and concentrations that either co-occur with observed adverse biological effects (Long and Morgan 1991) or have been established as interim sediment quality criteria by Ontario, Canada (Persaud, et al. 1993). The same mink dietary studies used to derive SW thresholds are used to derive site-specific thresholds for PCBs in sediment that protect mink.

The calculated site-specific surface water thresholds of 0.0016 and 0.00197 µg/L are used along with the mean site-specific sediment/surface water partition factor of 301,712 (rounded to 302,000) to derive site-specific sediment thresholds. Again, these sediment thresholds conservatively assume that mink diet is comprised of 100 percent fish and that the primary abiotic source of PCBs in mink prey is instream sediment. These mink-based PRGs are considered protective of riverine mink that consume fish. This approach for deriving mink-based sediment PRGs is justified for the following reasons:

- the terrestrial components of mink diet are minimal compared to aquatic components, represented by fish
- PRG calculation from dietary concentrations (as performed below) rather than dose is appropriate because the receptor species (mink) and the test species (mink) used to derive dietary thresholds is the same
- PRGs based on a diet comprised of both aquatic and terrestrial prey species requires that both sediment and soil PRGs be calculated simultaneously, resulting in an array of results.

The derivation of these sediment PRGs follow:

No Effect SED PRG

= No Effect SW threshold * SW-to-SED Partition Factor

= 0.0016 µg PCB/L * 302,000

= 483 µg PCB/kg sediment

= 0.5 mg PCB/kg sediment

Low Effect SED PRG

= Low Effect SW threshold * SW-to-SED Partition Factor

= 0.00197 µg PCB/L * 302,000

= 595 µg PCB/kg sediment

= 0.6 mg PCB/kg sediment

The calculated site-specific PRGs for PCBs in sediment, based on preventing fish tissue from containing more than 0.5 and 0.6 mg PCB/kg wet weight and site-derived BAFs from surface water, are 0.5 and 0.6 mg PCB/kg sediment.

These sediment PRGs can also be viewed using the BSAF approach. This is not an independent derivation because it is based on the same water-sediment-fish relationships described above. As presented on Table 4-8, the average site-specific BSAF, based on all fish species collected onsite, is 1.02. This alternative method of viewing this derivation is as follows:

No Effect SED PRG

= No Effect Fish Tissue Threshold/BSAF

= 0.5 mg PCB/kg wet weight whole body fish/1.02

= 0.5 mg PCB/kg sediment

Low Effect SED PRG

= Low Effect Fish Tissue Threshold/BSAF

= 0.6 mg PCB/kg wet weight whole body fish/1.02

= 0.6 mg PCB/kg sediment

Viewing these derivations using the BSAF approach allows simple estimations of whole body fish PCB concentrations from sediment PCB concentrations. Because the mean BSAF is nearly one (1.02), whole body fish PCB concentrations can be approximated by total PCB concentrations in sediment (SED * 1.02 = Fish).

Effects Data Sources (Surface Soil and Floodplain Sediments)

Similarly, accepted critical effects concentrations for chemicals in surface soils and floodplain sediments have not been developed solely for the protection of ecological receptors. As for sediment (streambed) contaminants, site-specific data are considered to be the most useful and appropriate for evaluating the potential toxicity of API/PC/KR surface soils and floodplain sediments. Such data are not, however, available, and three other approaches are used in the risk characterization phase of this ERA.

First, PCB concentrations in onsite surface soil and floodplain sediments are compared to background concentrations based on relevant and available data. Second, more general data sources on the potential hazards of contaminated surface soil and floodplain sediments are used to additionally evaluate the potential toxicity of API/PC/KR surface soil and floodplain sediment. Critical threshold levels for chemicals in surface soils, based on several soil functions including the protection of

wildlife, have been derived by and used in various countries (e.g., Norway; The Netherlands; West Germany; England; Ontario and Quebec, Canada) for several years (Siegrist 1989). The most appropriate critical threshold levels from sources such as these, based on general acceptance and data quality and quantity, are used to evaluate the potential toxicity of PCBs in surface soil and floodplain sediment. Evaluation of these alternative data sources suggests that the Ontario and Quebec (Siegrist 1989) values are the most appropriate and useful for this ERA. Preferred data (e.g., site-specific soil toxicity data) are unavailable, but the comparisons of PCB concentrations in onsite surface soil to threshold values (e.g., those derived by Ontario and Quebec) contribute to the weight-of-evidence regarding the potential toxicity of API/PC/KR surface soils and floodplain sediments. Because the soil threshold values presented in Siegrist (1989) and the sediment toxicity database of Long and Morgan (1991) are general and not site-specific, they can only contribute to multiple lines of evidence concerning the potential toxicity of surface soil or sediment. They are not, therefore, used alone to definitively describe API/PC/KR surface soil or floodplain sediment as toxic.

Media- and Receptor-Specific Dose-based TRVs

Media-specific and receptor-specific TRVs are calculated for a subset of representative receptors. These are dose-based NOAELs/No Effect Levels and LOAELs/Low Effect Levels for terrestrial species.

NOAELs and LOAELs are used as TRVs for red fox, great horned owl, muskrat, mouse, and mink. These TRVs form the basis for calculating hazard quotients and PRGs. Appendices C-2-A and C-2-B present the receptor-specific TRVs for all terrestrial and semi-aquatic receptors. As for mink, TRVs for non-raptor birds are based on the ED_x/EC_x approach introduced in Section 4.1 and discussed above (for mink). A discussion of the specific TRVs for non-raptor birds follows.

The no and low effect TRVs for non-raptor birds are based on a detailed analysis of the effects of PCBs on chicken, one of the best-studied and most sensitive avian receptors of the few species investigated to date. The TRVs are derived from exposure-response curves by interpolation of the effective dose to hens (ED_x) that corresponds to specific relative responses (calculated as the treatment response divided by the control response). The low effect dose is defined as 0.75 of the control response for a toxicological endpoint (ED_{25}, which represents a 25% decrease in response) and the no effect dose is equal to 90% of the control response (ED_{10}, which represents a 10% decrease in response).

The calculated low effect TRV for birds is 0.5 mg/kg_{BW}-d, based on Aroclor 1248, the predominant Aroclor detected in earthworms in the Kalamazoo River floodplain. The calculated no effect TRV for birds is 0.4 mg/kg_{BW}-d, also based on Aroclor 1248. TRVs calculated from exposure to commercial PCB products may underestimate the toxicity of PCBs in the field because of weathering and selective retention in biota. Effects may also be underestimated due to the relatively short-term exposure

durations of the majority of chicken studies (6 to 9 weeks). A single study continued exposure for 39 weeks in a single treatment, and this study showed increased adverse effects in the final weeks (Platonow and Reinhart 1973). However, since chickens are the most sensitive avian species tested to date with PCBs, application of uncertainty factors is not recommended for interspecific or subchronic-to-chronic extrapolations.

Appendix D presents a detailed summary of the ED_x/EC_x method used to derive TRVs for mink and non-raptor birds, and Appendices C-2-A and C-2-B present all the receptor-specific TRVs used to derive hazard quotients and PRGs.

Effects Data Sources (Bird Egg Data)

Bird egg data (Table 4-5b) are compared to egg-based thresholds for adverse effects (Table 4-9).

These effects data are based on relevant endpoints such as hatching success and survival of newly hatched young. Table 4-9 presents the selected bird egg toxicity or effects data used to estimate risks to bird eggs from PCB-contamination.

4.2.2 Stressor-Response Profiles

Stressor-response profiles (Table 4-10*)* present critical effects data for relevant ecological receptors or appropriate surrogate species that may be exposed to PCBs at the API/PC/KR site. The information presented in Table 4-10 includes relevant toxicity data from literature sources and includes site-specific information to the extent possible. For example, site-specific toxicity values for surface soil are included, along with a threshold streambed sediment PCB concentration, based on site-specific sediment/surface water partitioning, that is protective of aquatic species and piscivorous wildlife. These profiles include information on the lethal and sublethal effects that may be exhibited by exposed organisms correlated to media-specific PCB concentrations. Because effects and other relevant data are sparse for individual Aroclors, and because concentrations of detected PCBs (e.g., Aroclor 1260) approach concentrations of total PCBs measured, all effects data are based on total PCB concentrations. Likely responses to non-chemical stressors are not included in these profiles, but are qualitatively discussed below.

Siltation of Instream Substrate

Siltation, particularly as it contributes to the transport and deposition of PCB-containing residuals waste, may be contributing to ecological stress in the API/PC/KR area. Siltation can result in decreased dissolved oxygen concentrations, greater concentrations of contaminants sorbed onto fine grained sediments and other fine particulate matter, and shifts in macroinvertebrate community structure. For example, certain worm species and midge larvae are better adapted to silt than are stoneflies, caddisflies, and mayflies. Areas of siltation are likely to be characterized by lower species diversity than that found in areas of gravel/cobble. Siltation can directly (by smothering) and indirectly (by changing prey availability and community structure) affect survival of benthic macroinvertebrates. Siltation can adversely affect

fish reproduction and survival by smothering eggs and immature (prior to swim-up) fish. The paper waste residuals are very fine-grained particles which are easily suspended in the water column and when deposited concentrate PCBs in the sediments.

Impoundment Structures/Dams

Impoundment structures or dams can affect the movement of fish in the river, the distribution of PCBs and the exposure potential for aquatic receptors. Although impoundment structures present barriers to fish migration, the greatest threat from these structures is that they form a sink for the PCB residual materials. PCB residuals behind the formerly impounded areas are constantly being eroded into the Kalamazoo River and Portage Creek, and some of which will become bioavailable to aquatic receptors.

The impounded waters behind these structures provide excellent habitat for many game species and it is common to observe anglers at these locations. The exposure potential can be greater for both human and aquatic/terrestrial receptors at these sites.

Disturbed Terrestrial/Riparian Habitat

Most soil-dwelling animals, especially those that have limited mobility, are likely to avoid some terrestrial areas because preferred natural soils are no longer available when covered with significant amounts of contaminated sediments. While the potential toxicity of contaminated soils and streambank sediments cannot be ignored, it is likely that the physical presence of waste soils also affects habitat suitability for certain terrestrial organisms. Where terrestrial vegetation has either not been affected or has been re-established, a variety of terrestrial animals can find cover and food. Additionally, these disturbed areas are attractive sites for the development of "weedy" type plants, which can provide a food source for avian and terrestrial receptors.

4.2.3 Uncertainty Evaluation – Effects Assessment

In this section, the major sources of uncertainty in the effects analysis are identified and their potential impact on the ERA is evaluated. Media-specific toxicity data used in this ERA to describe the potential effects to ecological receptors are probably the primary source of uncertainty in the effects analysis.

Extrapolations are often used to relate measurement endpoints (e.g., lethal concentration) to assessment endpoints (e.g., macroinvertebrate abundance) or to relate one measurement endpoint (lethal concentration) to another (sublethal effects concentration). Extrapolations between taxa (e.g., species to species) or between responses (e.g., lethal to sublethal) are commonly used where specific data are limited. The use of these types of extrapolation is a commonly accepted practice but may increase uncertainty in risk assessment. The use of extrapolated data is, therefore, limited as much as possible in this ERA.

Data based on studies specific to the API/PC/KR area are preferred and are, therefore, used as much as possible in this ERA to minimize the uncertainties commonly associated with extrapolating toxicity or other data. Effects data for surface water and sediment contaminants are considered to be associated with low to moderate uncertainty, respectively. The unavailability of relevant site-specific surface water, sediment, and surface soil toxicity data increases uncertainty somewhat, but the availability of site-specific PCB concentrations in exposure media and resident biota helps minimize these uncertainties. There is considerably more uncertainty in the data used to evaluate the potential toxicity of contaminated surface soils because ecotoxicity data for terrestrial biota exposed to PCBs in surface soil are not as abundant as are data for evaluating PCBs in surface water and sediment.

As stated above, where possible, site-specific effects data are used to minimize uncertainty in the effects analysis. Because site-specific data are for the most part limited (to PCB tissue concentrations) or are unavailable (toxicity data), multiple lines of evidence are used to assess potential for ecological effects. This relies on ecological effects data from a large variety of appropriate and relevant data sources, and thus decreases the overall uncertainty compared to assessments based on only one or a few data sources. Several of the values used to quantitatively estimate critical threshold contaminant concentrations (e.g., AWQC, LOAECs, ED_{25}, site-specific tissue concentrations, Co-Occurrence Analysis (COA), Effects Range-Median (ER-M), and others) are often relatively similar in magnitude. These similarities allow greater acceptance of and support for each individual value, and in turn provide justification for using multiple lines of evidence in this ERA.

Figure 4-1
U95 Total PCB Concentrations
in Fish, SW, and SED

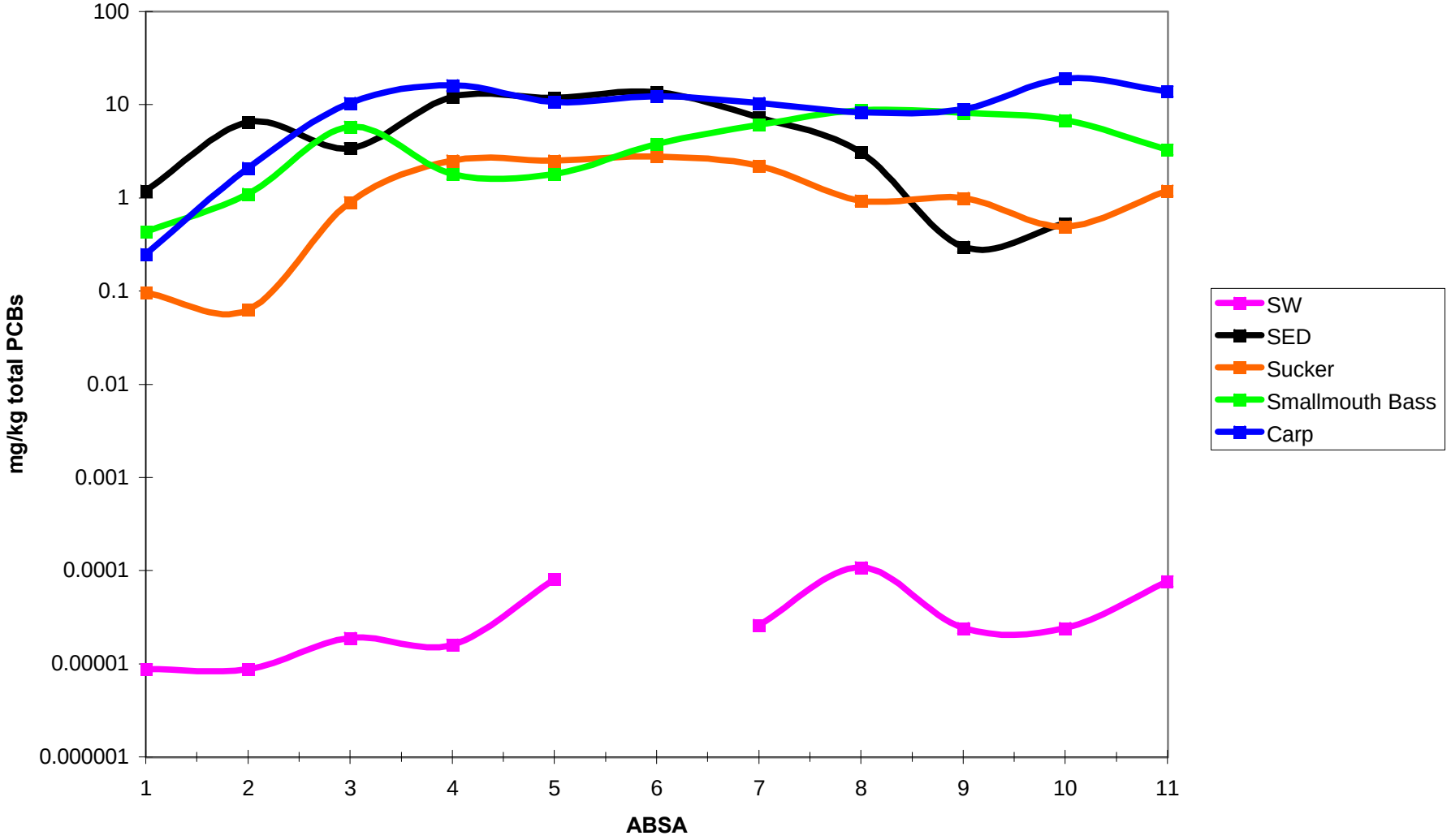

Fig4-1.xls

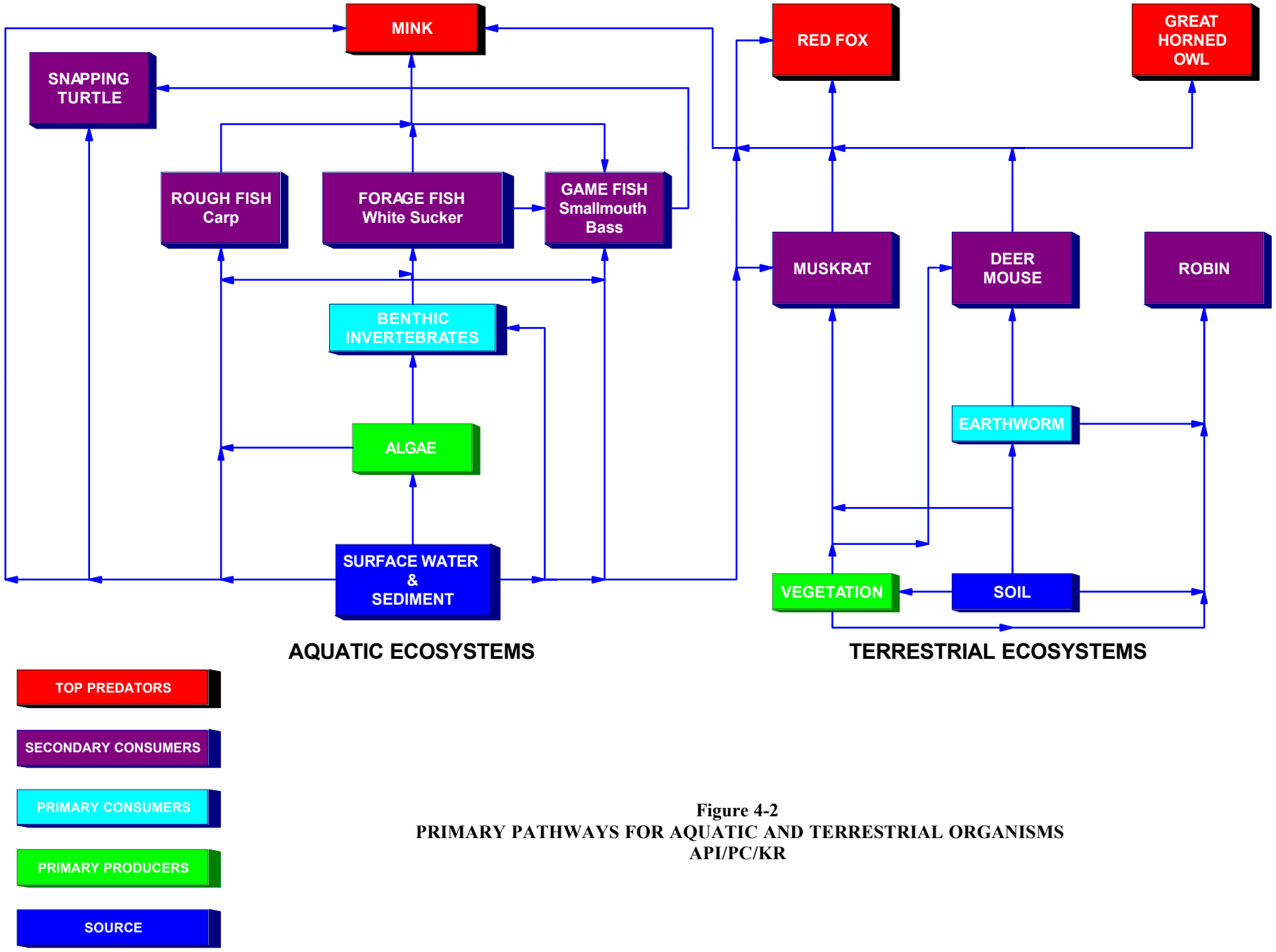

Figure 4-2
PRIMARY PATHWAYS FOR AQUATIC AND TERRESTRIAL ORGANISMS
API/PC/KR

Table 4-1
Sitewide Concentrations in Abiotic Media
API/PC/KR

Chemical	Abiotic Media	Concentration Range	
		Sitewide[1]	(reference area[2])
Aroclor 1016 Aroclor 1221 Aroclor 1232 Aroclor 1242 Aroclor 1248 Aroclor 1254 Aroclor 1260	The following media types were analyzed for individual Aroclors and Total PCBs: Surface Water (SW) Streambed Sediment (SED) Floodplain Sediment (FP SED) Surface Soil (SS)	Concentration range for individual Aroclors not applicable - ERA is focused on distribution and magnitude of Total PCBs	
Total PCBs	Groundwater (GW, µg/L) Surface Water (SW, µg/L) Streambed Sediment (SED, mg/kg) Floodplain Sediment (FP SED, mg/kg) Surface Soil (SS, mg/kg)	ND - 3 ND - 0.23 ND - 156 ND - 85 0.065 - 34.5	(NA) (ND) (NA) (NA) (ND) - 0.39

1 Sitewide: API/PC/KR except upstream reference area (ABSA 1)
2 Reference Area: ABSA 1
ND Non-detect
NA Data Not Available
Surface soil and FP SED data based on 0-6 inch depth

Table 4-2
Exposure Profile for PCBs - Chemical Properties
API/PC/KR

PCBs	Environmental Persistence	Bioconcentration Potential and Bioavailability
General	All PCBs are environmentally persistent, but less chlorinated Aroclors (e.g., 1016, 1221) are more easily degraded by bacteria than more chlorinated Aroclors such as Aroclors 1254 and 1260 (Eisler 1986).	Influenced by N-octanol/water partition coefficient (K_{ow}) which relates to solubility, and by stearic factors relating to chlorine substitution patterns (Eisler 1986). Bioaccumulation potential directly related to log Kow and stearic effects (Shaw and Connell 1982 in Eisler 1986). Generally, less chlorinated Aroclors are taken up to a lower degree than highly chlorinated Aroclors. An exception is found with Aroclor 1254, which apparently is taken up to a greater degree than all other Aroclors studied, including Aroclor 1260 (Eisler 1986). PCBs concentrate in liver, blood, and muscle in mammals. Generally, PCBs are lipophilic, and are most highly accumulated in fatty tissues. The pattern of Aroclor distribution in biological tissues, especially those of warm-blooded animals, only vaguely resemble the mixtures from which they originated (Hansen, et al. 1983 in Eisler 1986). Most commonly, PCBs measured in tissues are identified as Aroclor 1260. PCB metabolism and bioaccumulation is species-specific, and similar exposures result in different bioaccumulation rates.
Aroclor 1221	Persistent	Low to Moderate Bioaccumulation Potential/Bioavailability[1]
Aroclor 1232	Persistent	Moderate Bioaccumulation Potential/Bioavailability[1] Freshwater bioconcentration factor (BCF) for white sucker (*Catostomus commersoni*) equals 5,500 (Frederick 1975 in EPA 1980).
Aroclor 1016	Persistent	Moderate Bioaccumulation Potential/Bioavailability[1]
Aroclor 1242	Persistent	Moderate to High Bioaccumulation Potential/Bioavailability[1] Freshwater BCFs range from 36,000 (scud, *Gammarus pseudolimnaeus*, Nebeker and Pugilsi, 1974 in EPA, 1980) to 274,000 (fathead minnow, *Pimephales promelas*, Nebeker, et al. 1974 in EPA 1980).
Aroclor 1248	Persistent	High Bioaccumulation Potential/Bioavailability[1] Freshwater BCFs range from 52,000 (bluegill, *Lepomis macrochirus*, Stalling 1971 in EPA 1980) to 120,000 (fathead minnow, DeFoe, et al. 1978 in EPA 1980).
Aroclor 1254	Persistent	High Bioaccumulation Potential/Bioavailability[1] Freshwater BCFs range from 2,700 (phantom midge larvae, *Chaoborus punctipennis*, Mayer, et al. 1977 in EPA 1980) to 238,000 (fathead minnow, Nebeker, et al. 1974 in EPA 1980).
Aroclor 1260	Persistent	High Bioaccumulation Potential/ Bioavailability[1] BCF for fathead minnow equals 270,000 (DeFoe, et al. 1978 in EPA 1980)

[1] Estimated from degree of chlorination and available freshwater BCFs

Table 4-3
Exposure Information for Representative Ecological Receptors
API/PC/KR

Representative Receptor Group	Primary Stressor	Primary Potential Exposure Routes /Processes
Aquatic Plants (e.g., floating and rooted macrophytes and algae)	SW PCBs SED PCBs	SW Contact and Uptake SED/IWContact and IW Uptake
Aquatic Macroinvertebrates (e.g., mayfly larvae)	SW PCBs SED PCBS	SW Contact and Ingestion, Ingestion of PCB-contaminated Prey SED/IW Contact and Ingestion, Ingestion of PCB-contaminated Prey
Freshwater Game Fish (e.g., smallmouth bass)	SW PCBs SED PCBs	SW Contact and Ingestion, Ingestion of PCB-contaminated Prey SED/IW Contact and Ingestion, Ingestion of PCB-contaminated Prey
Freshwater Forage Fish (e.g., white sucker)	SW PCBS SED PCBs	SW Contact and Ingestion, Ingestion of PCB-contaminated Prey SED/IW Contact and Ingestion, Ingestion of PCB-contaminated Prey
Freshwater Rough Fish (e.g., common carp)	SW PCBs SED PCBs	SW Contact and Ingestion, Ingestion of PCB-contaminated Prey SED/IW Contact and Ingestion, Ingestion of PCB-contaminated Prey
Terrestrial Invertebrates (e.g., earthworms)	SS/FP SED PCBs	SS/FP SED Contact and Ingestion
Small Burrowing Terrestrial and Semi-aquatic Mammals (e.g., deer and white-footed mouse, muskrat)	SED/FP SED/SS PCBs	SED/FP SED/SS Contact and Ingestion, Ingestion of PCB-contaminated Vegetation/Prey
Small Omnivorous/Carnivorous Mammals (e.g., mink)	SW/SED/FP SED PCBs	Ingestion of PCB-contaminated Aquatic and Terrestrial Prey
Top Predators (e.g., red fox, great horned owl, bald eagle)	SW/SED/FP SED/SS PCBS	Ingestion of PCB-contaminated aquatic and terrestrial prey

SW	Surface Water
FP SED	Floodplain Sediment/Soil
IW	Interstitial Water
SED	Instream Sediment
SS	Surface Soil

Table 4-4
Potential Exposure via Contaminant Ingestion Pathway for Representative Aquatic and Terrestrial Organisms
API/PC/KR

Representative Receptor Group	Primary PCB Exposure Media	Discussion of Uptake/Ingestion Pathway
Aquatic Plants (e.g., floating and rooted macrophytes and algae)	SW SED	Hydrophobic PCBs in the water column are physically adsorbed on particulate matter, including algal cells (Eisler 1986). In addition, PCBs can be transferred from aqueous solution into algal lipids. These PCBs then can cause direct toxic effects to algae by inhibiting photosynthesis and motility. Finally, PCBs accumulated by algae are readily introduced into aquatic food chains (Rohrer, et al. 1982 in Eisler 1986).
Aquatic Macroinvertebrates (e.g., mayfly larvae)	SW SED	PCBs can be taken up by aquatic macroinvertebrates via ingestion of surface water, sediment, sediment pore water, and PCB-contaminated prey such as algae. Uptaken PCBs can cause direct toxic effects in macroinvertebrates, and can also be passed on to upper trophic level organisms through ingestion of PCB-contaminated macroinvertebrates. In addition, certain types of macroinvertebrates, such as mysid crustaceans in Lake Michigan, have a low assimilation efficiency for PCBs and a high efficiency for fecal excretion of ingested PCBs (Evans, et al. 1982 in Eisler 1986). PCB uptake from sediment by chironomids (midge larvae) can be correlated to sediment PCB concentration (Larsson 1984 in Eisler 1986). PCBs can be transported from aquatic to terrestrial environments via aquatic midge larvae to terrestrial midge adults (Larsson 1984 in Eisler 1986). Terrestrial consumers of adult midges can therefore be indirectly exposed to sediment-source PCBs.
Freshwater Game Fish (e.g., smallmouth bass)	SW SED PREY	More persistent and highly chlorinated PCBs can be found in trace amounts in fish from almost every major river in the United States (Schmitt, et al. 1985 in Eisler 1986). PCB-contaminated sediments and atmospheric deposition are the most important sources of PCBs in fish (Eisler 1986). Several studies reveal downward trends in PCB concentrations in whole body fish from throughout the U.S., especially for less chlorinated PCBs such as Aroclor 1242 (Eisler 1986). Total PCBs in fish measure environmental PCB contamination more reliably than do measurements for specific commercial mixtures such as Aroclor PCBs (Schmitt, et al. 1985 in Eisler 1986). Diet is major route of PCB uptake in most fish, but water can be a major source of PCB uptake in certain species under certain conditions (Greig, et al. 1983 in Eisler 1986). Although lipophilic, PCBs can also be deposited in gonads, eggs, muscle, and skin to varying degrees, depending on fish species (Eisler 1986).
Freshwater Forage Fish (e.g., white sucker)	SW SED	As above, but ingestion of prey less important because of omnivorous diet. Uptake of PCBs expected to be lower than for piscivorous gamefish or bottom dwelling rough fish.
Freshwater Rough Fish (e.g., common carp)	SW SED	As above, but ingestion of prey less important because of mostly herbivorous diet. Incidental ingestion of sediment may be important exposure route for bottom dwelling rough fish such as common carp. Direct contact with and ingestion of PCB-contaminated pore (interstitial) water may greatly increase exposure potential for benthic rough fish such as common carp.
Terrestrial Invertebrates (e.g., earthworm)	SS FP SED	Little data exist on PCB transfer from surface soil and floodplain sediments to earthworms. Earthworms have depurated ingested surface soil (i.e., “empty” earthworms) are expected to have higher whole body PCB concentrations than surface soils from which they were collected because of bioaccumulation.

Table 4-4
Potential Exposure via Contaminant Ingestion Pathway for Representative Aquatic and Terrestrial Organisms
API/PC/KR

Representative Receptor Group	Primary PCB Exposure Media	Discussion of Uptake/Ingestion Pathway
Small Burrowing Terrestrial and Semi-Aquatic Mammals (e.g., deer and white-footed mouse, muskrat)	SED FP SED PREY	Terrestrial burrowing rodents such as the white-footed deer mouse, are likely to ingest PCBs primarily through ingestion of invertebrate prey and plants. Vegetation portion of the diet is expected to contribute only small amounts of PCBs compared to contribution from animal prey. Semi-aquatic burrowing mammals like muskrats that are primarily herbivorous are most likely to take in PCBs through incidental ingestion of PCB-contaminated streambed and floodplain sediments. Omnivorous and herbivorous small mammals are expected to have lower PCB exposures than carnivorous species, especially those that consume substantial amounts of aquatic prey (e.g., mink).
Small Omnivorous/ Carnivorous Mammals (e.g., mink)	PREY	Mink are especially sensitive to PCBs, and their diet includes organisms that are most likely to be highly contaminated with PCBs (rough fish, benthic invertebrates such as crayfish, etc.). Several studies suggest that more highly chlorinated PCBs are eliminated more slowly than lower chlorinated PCBs in semi-aquatic carnivorous mammals studied (Eisler 1986). May be exposed via riverine diet, based predominately on fish, or via wetland diet, consisting of crayfish, muskrat, birds, and amphibians.
Top Predators (e.g., red fox, great horned owl, bald eagle)	PREY	PCB contamination most important to top predators (upper level carnivores) compared to lower trophic level organisms (Shaw and Connell 1982; Malins, et al. 1980 in Eisler 1986). Consumers of PCB-contaminated fish are likely to be at most risk because elevated PCB concentrations are expected in fish and other aquatic biota. Exposure through ingestion of prey must consider exposure frequency and duration as well as diet, and foraging range of top predators is critical to this evaluation.

Table 4-5a
Concentration and Distribution of Total PCBs in Sampled Biota and Abiotic Media
API/PC/KR

Media (ppm ww biota, dw abiotic)	TBSA 11 ABSA 1 reference	ABSA 2	Portage Creek	ABSA 3	TBSA 10 ABSA 4	ABSA 5	ABSA 6 Plainwell	ABSA 7 Otsego	TBSA 3, 5 ABSA 8 Trowbridge	ABSA 9	TBSA 1 ABSA 10 Allegan	ABSA 11
Smallmouth Bass [1]												
(max)	0.62	1.8		15	2.3	7.9	8.3	7.6	11	12	8.4	5.0
(mean)	0.35	0.83		3.6	1.4	4.6	2.5	5.1	6.9	6.5	5.6	2.6
(U95)	0.43	1.1		5.8	1.8	1.8	3.8	6.1	8.7	8.2	6.8	3.3
Sucker [1]												
(max)	0.14	0.8	2.4	1.0	2.9	3.1	4.6	2.8	1.1	1.7	0.92	1.6
(mean)	0.074	0.054	1.4	0.081	2.2	2.2	2.2	2.1	0.78	0.81	0.35	1.1
(U95)	0.096	0.063	1.9	0.90	2.5	2.5	2.8	2.3	0.93	1.0	0.49	1.2
Carp [1]												
(max)	0.41	4.2	10.8*	15	21	14	20	25	14	21	36	32
(mean)	0.20	1.4		8.1	12.8	8.8	8.5	6.3	6.5	5.6	13.2	8.9
(U95)	0.25	2.1		10.4	16.1	10.7	12.3	10.5	8.3	9.0	19.1	13.9
Terrestrial Plants												
(max)									0.069			
(mean)									0.023			
Earthworm [1]												
(WB max)	ND				0.66				3.2 (TBSA 3) 2.2 (TBSA 5)			
White-footed/ Deer Mouse[1]												
(WB max)	ND				0.28				0.45 (TBSA 3) 0.38 (TBSA 5)		0.35	
Muskrat[2]												
(WB max)	ND						0.6	0.2	2.9		1.1	
(liver max)	ND						0.7	0.3	1.2		0.5	
Mink[2]												
(WB max)	2.0						2.6	none	5.6		3.2	
(liver max)	1.5						2.4	collected	2.4		12.5	
Surface Water [3]												
(max)	0.0000075	0.0000075	0.000230	0.000048	0.000035	0.000091	no data	0.000071	0.000120	0.000052	0.000028	0.00012
(mean)	0.0000063	0.0000063	0.000058	0.000015	0.000013	0.000062	no data	0.000022	0.000075	0.000020	0.000018	0.000059
(U95)	0.0000088 (ABSA 1-2)	0.0000088 (ABSA 1-2)	0.000059	0.000019	0.000016	0.000081	no data	0.000026	0.000108	0.000024	0.000024	0.000077
Streambed SED[3]												
(max)	no data	2.4	120	86	44	100	94	156	91	7.2	0.73	1.4
(0-6≅) (mean)	no data	0.91	31.3	2.3	1.6	6.1	5.4	4.9	2.9	2.4	0.20	0.27
(U95)	no data	1.2	47.1	6.5	3.4	12.2	11.8	13.6	7.3	3.1	0.30	0.53

Table 4-5a
Concentration and Distribution of Total PCBs in Sampled Biota and Abiotic Media
API/PC/KR

Media (ppm ww biota, dw abiotic)	TBSA 11 ABSA 1 reference	ABSA 2	Portage Creek	ABSA 3	TBSA 10 ABSA 4	ABSA 5	ABSA 6 Plainwell	ABSA 7 Otsego	TBSA 3, 5 ABSA 8 Trowbridge		ABSA 9	TBSA 1 ABSA 10 Allegan	ABSA 11
FP SED[4]													Ottawa Marsh
(max)	no data	no data	no data	no data	no data	85	no data	36	81		no data	no data	0.04 - 2.8
(mean)	no data	no data	no data	no data	no data	10.9	no data	8.4	12.3		no data	no data	(x = 0.77)
(U95)	no data	no data	no data	no data	no data	16.2	no data	11.7	15.9		no data	no data	Potaw. Marsh
													0.04 - 1.97
													(x = 0.37)
									TBSA 3	TBSA 5			
Surface Soil[5]													
(max)	0.39	no data	no data	no data	10.2	no data	no data	no data	32.6	34.5	no data	0.23	no data
(mean)	0.21	no data	no data	no data	6.5	no data	no data	no data	24.5	25.1	no data	0.17	no data
(U95)	0.33	no data	no data	no data	8.9	no data	no data	no data	28.3	30.2	no data	0.23	no data
Mean Streambed SED/SW Partition Factor (Kd)[6]	301,712			342,105	212,500			523,077			129,167		

ND PCBs Not Detected
no data no recent data available for location or media type
NA Not applicable
* Estimated from filet and remaining carcass PCB concentrations (0.90 * PCB conc of remaining carcass: 0.90*12 mg/kg)

Footnotes:
1) Blasland, Bouck & Lee, Biota Investigation, July 1994.
2) MDNR, June 1994
3) Blasland, Bouck & Lee TM16, March 1995 (SW PC, ABSA 3,4,7,9,10) and TM10, April 1994 (SED ABSA 3,4,5,6,7,8,9)
Blasland, Bouck & Lee Description of the Current Situation, May 1992 (SED PC, ABSA 2, 10, 11 and SW ABSA 1,2,5,8, 11)
Surface Water Data for ABSAs 1 and 2 from samples taken at location near border of ABSA 1 and 2
Surface Water Data for ABSAs 1 and 2 estimated from two samples, less than detection limit, using half the detection limit
4) Blasland, Bouck & Lee, Former Impoundment Sediment and Geochronologic Dating Investigation, 1994, includes data analyzed in 1997 (ABSA 11 data from wetland sediments/soils)
Blasland, Bouck & Lee Description of the Current Situation, 1992 (ABSA 10, single sample)
5) Blasland, Bouck & Lee, Results of Phase I TBSA Soil Sampling, February 1994
6) K_d calculated only for ABSAs where reasonably synoptic (1993/1994) SED data were collected

Table 4-5b
Concentration and Distribution of Total PCBs in Bird Eggs
API/PC/KR

Species	PCB Conc (mg/kg)	Location	Year Collected	Collected/Analyzed by	Reference
RW Blackbird	1.64	Trowbridge Dam	1995	C. Mehne/A.D. Little Lab., Cambridge, Massachusetts	1
	1.61	Trowbridge Dam	1995	C. Mehne/A.D. Little Lab., Cambridge, Massachusetts	1
	0.0094	Ottawa Marsh, ASGA	1995	C. Mehne/A.D. Little Lab., Cambridge, Massachusetts	1
	1.77	Otsego Dam	1995	C. Mehne/A.D. Little Lab., Cambridge, Massachusetts	1
	1.05	Otsego Dam	1995	C. Mehne/A.D. Little Lab., Cambridge, Massachusetts	1
Robin	3.77	Plainwell Dam	1995	C. Mehne/A.D. Little Lab., Cambridge, Massachusetts	1
	0.405	Plainwell Dam	1995	C. Mehne/A.D. Little Lab., Cambridge, Massachusetts	1
GH Owl	22.46	Caulkin's Dam, ASGA	1995	C. Mehne/A.D. Little Lab., Cambridge, Massachusetts	1
	90.8	Koopman's Marsh, ASGA	1993	C. Mehne/Animal Health Diag. Lab., Lansing, Michigan	2
	15.94	High Banks Game Refuge, ASGA	1994	C. Mehne/Illinois Dep. of Agriculture, Centralia, Illinois	2
Wood Duck	0.736	Otsego Dam	1995	C. Mehne/A.D. Little Lab., Cambridge, Massachusetts	1
	0.265	Ottawa Marsh, ASGA	1994	C. Mehne/Illinois Dep. of Agriculture, Centralia, Illinois	1
	0.446	Ottawa Marsh, ASGA	1994	C. Mehne/Illinois Dep. of Agriculture, Centralia, Illinois	1
	0.315	Ottawa Marsh, ASGA	1994	C. Mehne/Illinois Dep. of Agriculture, Centralia, Illinois	1
	0.446	Ottawa Marsh, ASGA	1994	C. Mehne/Illinois Dep. of Agriculture, Centralia, Illinois	1
	0.373	Ottawa Marsh, ASGA	1994	C. Mehne/Illinois Dep. of Agriculture, Centralia, Illinois	1
GB Heron	1.48	ASGA, Ottawa Marsh	1993	C. Mehne/Animal Health Diag. Lab., Lansing, Michigan	2
	4.74	ASGA, Ottawa Marsh	1993	C. Mehne/Animal Health Diag. Lab., Lansing, Michigan	2
	7.67	ASGA, Ottawa Marsh	1993	C. Mehne/Animal Health Diag. Lab., Lansing, MichiganI	2
	2.30	ASGA, Ottawa Marsh	1993	C. Mehne/Animal Health Diag. Lab., Lansing, Michigan	2
	2.31	ASGA, Ottawa Marsh	1993	C. Mehne/Animal Health Diag. Lab., Lansing, Michigan	2
	44.38	ASGA, Ottawa Marsh	1993	C. Mehne /Animal Health Diag. Lab., Lansing, Michigan	2
Wood Thrush	1.93	Plainwell Dam	1995	C. Mehne /Animal Health Diag. Lab., Lansing, Michigan	2

Species	PCB Conc (mg/kg)	Location	Year Collected	Collected/Analyzed by	Reference
Yellow Warbler	1.31	Otsego Dam	1995	C. Mehne/A.D. Little Lab., Cambridge, Massachusetts	1
RT Hawk	2.31	High Banks Game Refuge, ASGA	1993	C. Mehne/Animal Health Diag. Lab., Lansing, MichiganI	2
	4.47	Caulkins Dam, ASGA	1994	C. Mehne/Illinois Dep. of Agriculture, Centralia, Illinois	2
	27.12	Ottawa Marsh, ASGA	1994	C. Mehne/Illinois Dep. of Agriculture, Centralia, Illinois	2
Bald Eagle	102.29	Ottawa Marsh, ASGA	1994	J. Marshall and C. Mehne/Mississippi State Chem. Lab, Mississippi State, Mississippi	2
	123.27	Ottawa Marsh, ASGA	1994	J. Marshall and C. Mehne/Mississippi State Chem. Lab, Mississippi State, Mississippi	3
	53.34	Highbanks Game Refuge, ASGA	1996	J. Marshall and C. Mehne/Mississippi State Chem. Lab, Mississippi State, Mississippi	2
	31.68	ASGA	1996	J. Marshall and C. Mehne/Mississippi State Chem. Lab, Mississippi State, Mississippi	2

References

1. Stratus Consulting Inc. 1999a. Laboratory Data Sheets and Chain of Custody Forms, Copies from D. Beltman, Stratus Consulting. Laboratory Data from 1995 Collection of Bird Eggs for PCB Analysis. Submitted to Camp Dresser & McKee Inc. (CDM) in September 1999.
2. C. Mehne 1994 in MDEQ, MDAG, USFWS, NOAA 2000 - Michigan Department of Environmental Quality (MDEQ), Michigan Department of Attorney General (MDAG), U.S. Fish and Wildlife Service (USFWS), and National Oceanic and Atmospheric Administration (NOAA). 2000. Notice of Intent to Perform an Assessment and Preassessment Screen. Kalamazoo River Environment Site, Michigan.
3. Letter from D. Best, USFWS, to S. Cornelius, MDEQ, 1996

Table 4-6
Measured Soil-to-Terrestrial Plant BAFs for PCBs
(garden plot data, ABSA 8, CDM 2000)
API/PC/KR

Soil PCB Conc. (mg/kg)	Plant PCB Conc. (mg/kg)	Plant Species	Soil to Plant BAF
3.33	0.0236	Peppers	0.0071
3.33	0.0415	Carrots	0.0125
3.33	<0.0025	Tomatoes	<0.0008[1]
3.33	0.0093	Rhubarb	0.0028
16.7	0.00318	Potatoes	0.00019
0.66 and 4.04	0.00931	Horseradish	0.008 (mean)
0.66 and 4.04	0.025	Cucumber	0.022 (mean)
0.66 and 4.04	0.0692	Lettuce	0.061 (mean)
Mean	**0.016**		
U95 BAF	**0.037**		

[1] 0.0008 used as BAF for fruits and berries in food chain modeling

Table 4-7
Literature-Based Soil-to-Terrestrial Plant BAFs for PCBs
API/PC/KR

Plant BAF	PCB Soil Application Rate	Receptor	Method	Reference
0	0.05, 0.5, 1 ppm	Carrot, radish	Experimental	Moza, et al. 1976 and Wallnofer, et al. 1975 in Pal, et al. 1980
0	Unknown	Mature tomato plants	Experimental	Wallnofer 1973 - 1974 (unpub) in Pal, et al. 1980
0.0008	**-**	**Green tomatoes (represents fruit/berries)**	**Measured, Co-Located Soil and Plant**	**CDM 2000**
0.002	100 ppm	Soybean sprouts	Experimental	Suzuki 1977 in Pal, et al. 1980
0.01	0.3 ppm	Sugarbeet leaves	Experimental	Wallnofer, et al. 1975 in Pal, et al. 1980
0.015	–	Aboveground vegetation	Theoretical, log TF=1.588-log (K_{ow})	Travis and Arms 1988
0.016	0 - 1,000 ppm	Soybean	Experimental	Weber, et al. 1979 in Pal, et al. 1980
0.03	0.17 B 0.24 ppm	Sugarbeet leaves	Experimental	Moza, et al. 1978b in Pal, et al. 1980
0.04	**–**	**8 Crop Species (all tissues)**	**Measured, Co-Located Soil and Plant**	**CDM 2000**
0.07	0.17 B 0.24 ppm	Sugarbeet roots	Experimental	Moza, et al. 1978b in Pal, et al. 1980
0.16	100 ppm	Carrot roots	Experimental	Iwata, et al. 1974 in Pal, et al. 1980
0.16	0.05, 0.5, 5 ppm	Carrot roots	Experimental	Wallnofer, et al. 1975 in Pal, et al. 1980
0.17	0 - 1,000 ppm	Fescue	Experimental	Weber, et al. 1979 in Pal, et al. 1980
0.25	1 ppm	Carrot roots and leaves	Experimental	Moza, et al. 1976 in Pal, et al. 1980
0.5	0.3 ppm	Sugarbeet whole plant	Experimental	Wallnofer, et al. 1975 in Pal, et al. 1980
0.80	0.17 B 0.24 ppm	Weeds	Experimental	Moza, et al. 1978b in Pal, et al. 1980
0.96	1 ppm	Weeds	Experimental	Moza, et al. 1976 in Pal, et al. 1980
1.3	1 - 2 ppm	Fresh plant B barley	Mean of measured conc in plant/mean measured conc in soil	Trapp, et al. 1990

Table 4-8
Calculated Aquatic BCFs[1]/BSAFs [1] and Terrestrial BAFs[1] for Representative Food Web Species (based on primary exposure media)
API/PC/KR

Location	SM Bass BAF (SW)	SM Bass BSAF (SED)	Sucker BAF (SW)	Sucker BSAF (BSAF)	Carp BAF (SW)	Carp BSAF (SED)	Earthworm BAF[2] (SS)	White-footed/Deer Mouse BAF (SS)
ABSA 3	305,000	0.9	47,000	0.1	547,000	1.6		
ABSA 4 TBSA 10	113,000	0.5	156,000	0.7	1,000,000	4.7	0.07	0.03
ABSA 5	NA	0.1	NA	0.2	NA	0.9		
ABSA 6	NA	0.3	NA	0.2	NA	1.0		
ABSA 7	235,000	0.4	88,000	0.2	404,000	0.8		
ABSA 8 TBSA 3, 5	NA	1.2	NA	0.1	NA	1.1	0.113 (TBSA 3) 0.073 (TBSA 5)	0.016 (TBSA 3) 0.013 (TBSA 5)
ABSA 9	342,000	2.6	42,000	0.3	375,000	2.9		
ABSA 10/TBSA 1	NA	NA	NA	NA	NA	NA	0.109	1.52
Average	249,000	0.88	83,000	0.28	583,000	1.9	0.09	0.40
Average FISH BAF = 305,000 Average FISH BSAF = 1.02								

[1] BCFs/BAFs based on U95 PCB Conc (biota)/U95 total PCB Conc (exposure media) Data from Table 4-5a. Values are derived only for locations where reasonably synoptic data were collected

Values are rounded to the nearest one thousand. SW: Surface Water SED: Instream Sediment SS: Surface Soil/Floodplain Sediment from TBSAs

[2] Worm BAFs based on depurated carcass (measured).

NA: Not Applicable because 1) media quality and/or biological data not collected or 2) PCBs were not detected in sampled biota.

Table 4-9
Adverse Effects Associated with Bird Egg PCB Concentrations
API/PC/KR

Species	Egg PCB Conc (mg/kg)	Effect	Reference
Chicken	0.36	NOAEC, egg hatchability	Scott 1977 in 2
	0.95	NOAEC, egg hatchability	Britton and Huston 1973 in 2
	1.5	LOAEC, egg hatchability	Britton 1973 in 1
	2.5	LOAEC, egg hatchability	Scott 1977 in 1
	2.8	**Mean NOAEC**	Calculated, N = 4
	3.0	egg hatchability	Brunstrom 1988 in 1
	4.0	LOAEC, deformities and egg hatchability	Tumasonis, et al. 1973 in 2
	4.8	egg hatchability	Lillie 1975 in 1
	<5.0	NOAEC, egg production and female fertility	Platonow and Reinhart 1973 in 2
	5.0	LOAEC, egg production and femaile fertility	Platonow and Reinhart1973 in 1 and 2
	6.2	**Mean LOAEC**	Calculated, N = 6
	5.0	NOAEC, egg hatchability and 2-fold increase in deformities	Summer et al. 1996 a,b
	24	LOAEC, egg hatchability and 2-fold increase in deformities	Summer et al. 1996 a,b
Ring-necked Pheasant	1.0 1.8	egg lethality egg hatchability	Brunstrom 1986 in 1 Dahlgren 1972 in 1
	16	egg lethality	Peakall 1972 in 1
Tree Swallow	5.7	LOAEC, reproductive behavior	McCarty and Secord 1999 in 2
Herring Gull	5	Egg hatchability	Ludwig 1993 in 1
Foster's Tern	4.5	NOAEC, hatching success	Kubiak, et al.1989 in 2
	7.0	NOAEC, population size or reproductive success	Bosveld and Van den Berg 1994 in 2
	19.0	LOAEC, population size or reproductive success	Bosveld and Van den Berg 1994 in 2
	22.2	LOAEC, egg lethality	Kubiak, et al.1989 in 1
Caspian Tern	4.2	LOAEC, egg hatchability	Yamashita 1993 in 1
Double-crested Cormorant	3.5	egg hatchability	Tillitt 1993 in 1
Bald Eagle	1.5	NOAEC (est. from mean LOAEC/10)	Calculated, LOAEC N = 5
	4.0	LOAEC, egg lethality	Kubiak 1991 in 1
	4.0	LOAEC, population size or reproductive success	Ludwig et al. 1993 in 2
	4.5	LOAEC, 40% decrease in productivity	Wiemeyer 1984
	7.2	NOAEC, "successful" nests	Wiemeyer et al. 1984
	7.7	Mean LOAEC	Calculated, N = 5
	13	LOAEC, "unsuccessful" nests	Wiemeyer et al. 1984
	13	LOAEC, population size or reproductive success	Bosveld and Van den Berg 1994 in 2

1: RCG/Hagler, Bailly, Inc. 1994
2: Stratus Consulting 1999b

Table 4-10
PCB Stressor-Response Profiles
API/PC/KR

Chemical Stressor	Media of Concern	Measurement Endpoint Concentrations	Measurement Endpoint Data Data Type/Species/Effects	References
Total PCBs (µg/L)	SW	0.00012	Wildlife Protection Criterion for Surface Water - Michigan	Act 451 1994, Part 4
		0.0016	Site-specific value to protect mink. Based on mean site-specific BAF for fish (305,000) and dietary no effect concentration for mink (0.5 mg/kg).	See text
		0.00197	Site-specific value to protect mink. Based on mean site-specific BAF for fish (305,000) and dietary low effect concentration for mink (0.6 mg/kg).	See text
		0.014	Chronic Ambient Water Quality Criterion	EPA 1980
		0.14	Lowest chronic value, freshwater aquatic plants	Suter and Tsao 1996
		0.2 – 9	Range of chronic values (mean of ranges) for Aroclors 1242-1260, fathead minnow	EPA 1980
		0.8 – 15	Range of chronic values (mean of ranges) for freshwater invertebrates	EPA 1980
Total PCBs (mg/kg) (Aquatic/Semi-aquatic/Wetland)	SED FP SED	0.0029	Freshwater Screening Level Concentration (SLC)	Long & Morgan 1991
		0.01	No Effect Level, benthic organisms, Ontario	Persaud, et al. 1993
		0.054 – 3.1	Range of apparent effects concentrations (AET), multiple species	Long & Morgan 1991
		0.07	Lowest Effect Level, benthic organisms, Ontario	Persaud, et al. 1993
		0.1	Carp-based values based on GLI default values to protect mink	See Table 5-5 (MDEQ-SWQD)
		0.37	Concentration at which adverse effects are always observed	Long & Morgan 1991
		0.4	Effects Range-Median (ER-M)	EPA 1988b see text, EP approach*
		0.5	Calculated value to allow IW to remain below site-specific no effect SW threshold (0.0016 µg/L)	EP Approach/ Site-specific
		0.6	Calculated value to allow IW to remain below site-specific low effect SW threshold (0.00197 µg/L)	EP Approach/ Site-specific
		3.5	Calculated value to allow IW to remain below chronic AWQC (theoretical Kd)	EP Approach
		4.2	Calculated value to allow IW to remain below chronic AWQC (site-specific Kd: 302,000)	EP Approach

Table 4-10
PCB Stressor-Response Profiles
API/PC/KR

Chemical Stressor	Media of Concern	Measurement Endpoint Concentrations	Measurement Endpoint Data Data Type/Species/Effects	References
Total PCBs (mg/kg) (Terrestrial/upland)	FP SED SS	0.1	"A" concentration (background pollution), Quebec	Siegrist 1989
		1	"B" concentration (threshold), Quebec	Siegrist 1989
		6.5 – 21	Range of no effect PRGs (API/PC/KR-specific) to protect terrestrial / upland receptors (lowest value for robin)	See text
		10	"C" concentration, (contaminated), Quebec	Siegrist 1989
		8.1 – 63	Range of low effect PRGs (API/PC/KR-specific) to protect terrestrial / upland receptors (lowest value for robin)	See text

SW: Surface Water SED: Sediment FP SED: Floodplain Sediment/SS: Surface Soil
Equilibrium Partitioning approach (SED CONC=KD*IW CONC), (Site-specific: mean Kd=302,000, IW CONC = Chronic AWQC (0.000014 mg/l)

(Theoretical): SED CONC (mg/kg) = KD*IW CONC (mg/L)
KD = Koc * Foc
Foc = 0.084 (sitewide mean Foc)
KD = 2,944,422 * 0.082 = 247,331
log Koc = 0.937 log Kow - 0.006 (EPA Foc 1988b) = 6.469 (Koc = 2,944,422)
Mean log Kow (Aroclor 1260) = 6.91 (EPA 1988b)
SED CONC (mg/kg) = KD*IW CONC (mg/L
3.5 mg/kg = 247,331* 0.000014 mg/L